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Drinking Water Criteria Document for Man-Made Radionuclide Occurrence (MMRD) (Draft)

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Health Effects Branch
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I. SUMMARY

In preparing health criteria documents for drinking water the U.S. Environmental Protection Agency normally considers the effects of each individual chemical residue. However, for the approximately 200 man-made radionuclides a different approach is necessary and possible. While for many individual radionuclides knowledge concerning their health effects are minimal, a considerable understanding of the mechanisms of deposition of representative ones in the human body and the effects of different types of radiation on different tissues have been accumulated over the las 60 years. Thus even though each radionuclde/element has different adsorption and retention properties and decay schemes, both national and international groups of scientists have abeen able to develop models to calculate dose and the human health effects. These models are based on a knwledge of transfer of different radionuclides into and within the body, the biological half-life of the stable element, and the relationships between the type and energy of the radiation and its effects. The analysis presented in this document therefore relies heavily on the complex dosimetric model developed by the Internatioal Commission on Radiation Protection (ICRP).

This background document is unlike the other Health Criteria Documents developed by the Office of Drinking Water. The man-made radionuclides described here include approximately

200 isotopes. These radionuclides have different toxicokinetics and resultant health effects. For many of the radionuclides, the existing knowledge concerning health effects is minimal. Thus in order be able to handle this diverse set of radionuclides the analysis here relies on a dosimetric model (from ICRP Report 30). The tables in this document estimate the concentration of each isotope that can produce a dose equivalent of 4 rem/year whole body radiation. The calculation assumes an average lifetime of 70.7 years for the cohort of 100,000 used in the analysis and a risk of 200 excess fatal cancers in the cohort from a chronic uniform whole body dose rate of 0.1 mrem/year (computed as the geometric mean of the risk estimates based on the absolute and relative risk models). At a uniform whole body dose of 4 mrem/year, there are approximately eight excess cancers in the cohort of 100,000 people.

II. INTRODUCTION

A. Epidemiological and Experimental Approaches and Their Limitations

In the establishment of limits for contaminants in drinking water, the quantification of toxicological effects and other health risks associated with the particular contaminants must be established. Obviously, limits must be established that are below the threshold for acute toxicity and other non-stochastic effects associated with the impairment of the function of affected tissues or organs. For chemicals, threshold limit values have been established from animal studies and human exposure data when available. Similar studies and data have been used to establish threshold doses and non-stochastic limits for radiation (ICRP 1977a) that are deemed to prevent these effects.

With respect to the establishment of drinking water standards, the most difficult task, by far, is the estimation of the risk or probability of cancer or of other stochastic effects associated with a given contaminant level in a drinking water supply serving a given population. The difficulty of this task is due to the fact that cancer is a relatively prevalent disease that may be initiated by exposure to many natural and man-made agents, including chemicals, radiation and other factors. Thus, the incremental risk that might be associated with a given contaminant level at or near the standard can only be roughly estimated from extrapolations involving animal or human exposures at much higher levels. Obviously, when a substance is known to have mutagenic and carcinogenic properties and when no clear threshold can be established for the somatic and genetic effects of concern, no absolutely safe level of the substance can be established. This situation requires that the benefits to society from the technologies that give rise to the substance outweigh the risks, that the costs and other risks associated with alternative technologies make them less beneficial, and that exposures to the substance be maintained at levels as low as reasonably achievable (ALARA)(ICRP 1985).

Although all of these considerations are important in the establishment of the final values chosen for drinking water standards, the first step is the estimation of the risk associated with a given level of the contaminant. This report is limited to this very first step: the assessment of the health risks from exposure to radioactivity in drinking water and, specifically, of the carcinogenic risks.

Unlike certain man-made chemicals that might not occur naturally, radiation, as a cancer-initiating agent, has large and variable naturally-occurring components, including both external and internal sources, e.g., naturally-occurring radioactivity in drinking water, natural airborne radioactivity and external radiation from terrestrial and cosmic sources

(NCRP 1975, 1977, 1984a,b, ICRP 1984, Cothern et al. 1985). Although animal experiments and epidemiological studies involving human exposures can be and have been useful in the establishment of thresholds for non-stochastic effects and in the estimation of risks of cancer and hereditary effects, radiation protection standards for exposures of both workers and the public have been derived in recent years on the basis of dosimetric models that are based upon the available experimental and epidemiological information. The useful information has generally been limited to the high-dose range, well above the level that would be considered for the establishment of standards. Most scientific studies of the available information on the effects of radiation conclude that the risk or probability of cancer and hereditary effects increases with the radiation dose (ICRP 1977a, NAS 1972, 1977, 1980, UNSCEAR 1977, 1982).

Without specific useful information in the low-dose range of interest, it is generally thought to be prudent to assume that such effects have a dose threshold of zero. Also in general the greater the dose, the greater the effect on the individual or on the population exposed. Under this assumption, dose limits have been tacitly used as surrogates for the risks of the associated stochastic effects. A dosimetric approach using acceptable estimates of risks, therefore, is considered to be a reasonable method of establishing limits for radionuclides in drinking water. This method has the advantage of providing a consistent approach, regardless of the specific radionuclides, even though (1) many simplifying assumptions are first required to estimate the doses to all significantly irradiated organs and tissues and (2) an appropriate dose-risk model, with all of its simplifying assumptions, must then be used to provide an estimate of the associated risks. This approach makes the tacit assumption that the effects of concern can be related to the doses, i.e., the average specific radiation energy absorbed by organs and tissues, weighted by appropriate factors to reflect the biological effectiveness of different radiations and the sensitivity to the stochastic effects of concern. This approach, rather than that involving direct experimental or epidemiological evidence, is used here to establish the risk associated with radionuclides in drinking water.

Despite the fact that many simplifying assumptions are necessary for the use of this dose-risk criterion for radionuclides in drinking water, it has advantages over experimental or epidemiological approaches. The stochastic effects of concern may have many causes and may have a high frequency. This may result from man-made or naturally occurring initiating agents, . Experimental or epidemiological approaches generally do not have the sensitivity or accuracy necessary (in the presence of the background and variance associated with the given stochastic effect of interest) to quantitatively resolve the effects due to the agent of interest from those due to all agents that might initiate the stochastic effect under study. The absence of any statistically significant increase in a stochastic effect of interest from a given exposure level of some agent, such as radiation, does not mean that there is no effect. In light of the conservative assumption of a zero threshold for radiation-induced stochastic effects, such a conclusion would be invalid and could lead to inadequate standards.

B. Discussion

1. Internal Radiation Dose Recommendations in ICRP Publication 2 (ICRP 1960) and NCRP Report No. 22 (NCRP 1959)

The International Commission on Radiological Protection (ICRP) and the National Council on Radiation Protection and Measurements (NCRP) provided comprehensive recommendations for internal radiation protection in the publications cited above. Although both of these publications were limited to recommended limits for the exposures of workers, the concentration limits given for air and water for continuous occupational exposure were used to establish limits for the public by dividing these values by ten. In essence, the concentration limits provided an Annual Intake Limit (AIL) which, for standard man, would limit the integrated dose to the critical organ in the next 50 years (i.e., the committed dose) to the Maximum Permissible Annual Dose (MPAD) established for external radiation alone. The so-called "critical organ" was that organ or tissue that was expected to receive the largest committed dose relative to its established MPAD. Thus, the concentration limits established for the public correspond to one-tenth of the AILs and MPADs for workers. These 1959 recommendations of ICRP and NCRP, most of which are incorporated in the current NRC 10 CFR 20 regulation, were thus considered to limit the potential dose to an acceptable level of risk.

Although the Maximum Permissible Concentrations (MPCs) were intended to limit the committed doses and risks to an acceptable level, they were actually derived for convenience on the basis of an occupational, continuous intake model. When standard man has a continuous intake of 1 AIL/year for 50 years, at the end of this period, the critical organ will contain an amount equal to the Maximum Permissible Organ Burden (MPOB), which causes it to receive at this time a dose equivalent rate equal to 1 MPAD/year. When standard man has a single acute intake of 1 AIL, the critical organ will receive a dose over the next 50 years (i.e., committed dose) that is equal to 1 MPAD. This fact, which applies regardless of the mathematical form used to describe the retention of the radionuclide and any of its progeny, was the basis of ICRP's recommendation for limiting the intake in each year of practice to 1 AIL. It is not generally recognized that the ICRP 2 parameter values (e.g., the effective energy term, the biological half-life and the fraction of the body burden in the critical organ) generally apply only at the end of the chronic occupational exposure period of 50 years or to the calculation of the 50-year committed dose following a single acute intake. They were not intended to be applied over shorter time periods, e.g., 13 weeks or a year. This limitation is also true of current ICRP models and parameter values in ICRP Publication 30.

Because of the lack of specific metabolic and other information at the time, parameter values for radioelements in the ICRP Publication 2 were sometimes derived from the known distribution of stable elements in the diet and throughout various organs and tissues in the body. For example, the biological half-life of an element in a given organ or tissue of standard man was determined by assuming homeostasis and setting the biological elimination rate equal to the deposition rate of the stable element in the organ. This deposition rate was estimated from the product of the ingestion rate of the stable element in the diet, the fractional uptake to blood and the fractional deposition from the blood into the reference organ. It was felt that this procedure of treating each organ as a mutually exclusive compartment, which conceptually neglects the recycling of elements between cells and extracellular fluid, would yield a reasonable estimate of the effective mean residence time of a radioelement in a given organ or tissue and of the long-term content after 50 years of chronic occupational exposure. Because of the simple mathematical relationship between the dose rate for continuous exposure and the committed dose following a single acute exposure, this ICRP model would also be expected to give a reasonable estimate of the 50-year committed dose following a single acute intake. It would not, however, be generally appropriate for estimating doses over shorter time periods. Except for the fact that chosen parameter values were sometimes based upon the known distribution of stable elements taken into the body from the diet, which may not adequately describe either the uptake from the gastrointestinal (G.I.) tract or the systemic mean residence time of radioelements taken into the body in different chemical forms, the approach used in the ICRP Publication 2 was a reasonable one to use for the derivation of the MPCs.

New information on the biological effects of radiation, the metabolism and retention of radionuclides in the body and the anatomical and physiological models for reference man (ICRP 1975) and better data on radioactive decay schemes (ICRP 1983) have accumulated since the issuance of the ICRP Publication 2 in 1959. These factors and changes in the basic recommendations of ICRP, embodied in ICRP Publication 26, have led Committee 2 of ICRP to issue a new report for internal radiation protection. These most recent internal radiation dose recommendations are given in ICRP Publication 30, "Limits for Intakes of Radionuclides by Workers", in three parts (ICRP 1979, 1980, 1981).

2. ICRP Publication 30 Models and Internal Dose Recommendations

The ICRP Publication 30 replaces the recommendations in ICRP Publication 2. The derived limits include Annual Limits on Intake (ALIs) and Derived Air Concentrations (DACs) for occupational exposure. These limits extend the 1959 ICRP recommendation of limiting intakes so as to limit the committed dose to a single critical organ. By taking into consideration all significantly irradiated target organs and tissues and by weighting their committed dose equivalents by their risk weighting factors for cancer and hereditary effects, intakes for workers are limited in each year of practice so as to limit the committed effective dose equivalent to 5 rem for all organs. The risk weighting factors are those given in ICRP Publication 26. The weighting factor for a particular organ or tissue represents the proportion of the total lifetime stochastic risk that is due to that tissue when the whole body is irradiated uniformly. The total stochastic risk is calculated on the basis of cancer mortality and hereditary effects in the first two generations. For the purpose of establishing the derived limits, which consist of the ALIs and DACs, and for the purpose of controlling the combined external and internal committed risk which results from exposures in any year of practice to an acceptable level, a 5-rem committed effective dose equivalent from internal radiation is deemed to have the same lifetime stochastic risk as a uniform 5-rem irradiation of the whole body from external radiation. By limiting the combined external and internal committed effective dose equivalent to 5 rem in any year of practice, which under ALARA (as low as reasonable achievable-the concept of feasibility widely considered in the development of health physics standards) programs has resulted in average annual doses of less than 0.5 rem, the total stochastic risk over an occupational lifetime is comparable to that associated with safe industries and thought to be acceptable to workers (ICRP 1977b).

The recommendation in ICRP 30 that limits the combined external and internal dose equivalent to any single organ or tissue to 50 rem so as to prevent non-stochastic effects is very much akin to the ICRP Publication 2 recommendation limiting the committed dose equivalent to a single critical organ.

With respect to the exposure of individual members of the public, ICRP states that an annual combined external effective dose equivalent limit of 5 mSv (0.5 rem) is likely to result in average dose equivalents of less than this limit. Except for differences in organ size and in the metabolic characteristics of children compared to reference man, controlling the intakes of members of the public to one-tenth of the stochastic-based intake limits calculated for workers is expected to control doses, in the absence of any external radiation, to the 0.5-rem limit.

Although concentration limits are no longer given by ICRP for water, the occupational stochastic-based intake limits could be used to obtain derived water concentration limits for members of the public. Instead of using this simple approach, a separate dose-risk model, based upon many of the ICRP models and assumptions, is used here to estimate the risks associated with given levels of radionuclides in water. This approach is taken to more directly account for the risks of dying from competing causes and to establish a methodology that will be able to accommodate differences in ages and sex, should such differences be shown to be important for certain radionuclides and critical groups.

The US Environmental Protection Agency's Office of Drinking Water develops the dosimetric model basis for a standard for man-made radionuclides using the ICRP 30 dosimetric model as modified by the lifetable approach(Sullivan et al, 1981). As part of this development, the drinking water concentration and resulting effective dose equivalent levels are calculated for different lifetime risk levels. An average lifetime of 70 years is used.

3. Assumptions and Limitations of ICRP 30 Metabolic and Dosimetric Models

In order to indicate the applications and limitations of the metabolic and dosimetric models used here for ingestion intake, the major simplifying assumptions used in the ICRP Publication 30 are summarized as follows. These ICRP 30 models are applicable to so-called reference man (see ICRP publication 23) and are used here with some modifications. The modifications do not account for differences in age or sex, although such modifications, in principle, could be taken into account if appropriate parameter values were available. These considerations are discussed in the ICRP statement included with the ICRP Publication 39 (ICRP 1984), as well as in the ICRP Publication 30 itself.

a. Assumptions concerning chemical and physical form. In the ICRP Report 30, an effort was made to account for differences in the chemical and physical forms of radionuclides. With respect to ingestion, one of the most important parameter values affecting internal radiation doses is the fractional absorption from the G.I. tract to the blood. Different values for this fractional uptake, depending upon chemical form, are given in the ICRP Publication 30. As indicated in the report, radioelements that are present in the diet in organically complexed forms often have a higher uptake than do inorganic forms. Obviously, the chemical compound form also could influence the systemic retention and internal radiation doses following the uptake of radionuclides. For this reason, some results are given in this report for radiocompounds that exhibit a large variation in the fractional uptake from the G.I. tract.

b. Assumptions concerning systemic metabolic models. The metabolic models used to describe the systemic retention of elements are described in the Metabolic Data (ICRP 30) for each element. In general, specific chemical compounds are not considered. In some cases, the metabolic model may be quite detailed and may be supported by much experimental evidence from observations on humans. In other cases, data may be limited to a few observations of a single species of experimental animal. The ICRP indicates that this lack of data represents the largest factor of uncertainty in most estimates of the committed dose. Other assumptions and limitations that should be noted include:

(1) Uptake and clearance of a pseudo transfer compartment. For mathematical convenience, uptake is modeled into a pseudo transfer compartment. This compartment is deemed to be cleared via all pathways with a half-clearance time of six hours, unless otherwise specified in the metabolic data for the particular element. Because an empirically determined systemic whole body uptake retention function includes the systemic circulation, the addition of this pseudo compartment adds an artificial mean residence time of about nine hours to the systemic retention. This could have an impact on dose calculations and the distribution of dose within the body, especially for highly transportable radioelements that have

radioactive half-lives comparable to or less than the six-hour clearance half-time for this pseudo transfer compartment. For this reason, this transfer compartment was eliminated in the calculations performed here by assuming instantaneous deposition into the various source organs and tissues named in the metabolic models.

(2) Direct excretion from organs and tissues named in the metabolic models and organ retentions. The depiction of direct excretion from organs and tissues of deposition, although seemingly incorrect, is mathematically correct if the systemic whole body retention function is a true uptake retention function.Thus it must incorporate the dynamic process of recycling of the element between extracellular fluid and various tissues and organs of deposition. For example, in the many cases in which the systemic whole body uptake retention function is expressed by a sum of exponential terms with constant coefficients, it is correct to consider each exponential term as a pseudo one-way catenary compartment, in which the coefficient of each exponential term gives the effective fraction of an uptake that instantaneously deposits in that compartment. In each case the rate constant in the exponential term gives the effective fraction removed per unit time from that compartment by excretion.

The problems in the ICRP 30 model description are related to the facts that specific excretion pathways are not identified. Individual exponential terms in the systemic whole body retention function are often identified with structured physiological compartments when they cannot, in reality, be associated with any one specific physiological compartment if significant systemic recycling of the element occurs. The ICRP has recognized the problem regarding excretion and has stated that because of this assumption, doses to organs and tissues in the excretion pathways are not usually taken into account. In most cases, these doses will be small and will not contribute significantly to the committed effective dose equivalent.

Regarding the distribution or assignment of individual exponential terms of the systemic whole body uptake retention function among structured physiological compartments that are named in the ICRP 30 metabolic model for an element, a proper estimate will be made of the committed dose equivalent distribution in the body, provided that the distribution coefficients are chosen to reflect the long-term content. This distribution, however, will not normally describe the actual short-term retention in an organ or tissue, and it probably will not give a proper estimate of the dose distribution over shorter periods of time. Despite this problem and lacking more specific information, doses are calculated.

(3) Assumption of instantaneous uniform mixing in a

compartment. This assumption influences the distribution of the radionuclide content within an organ compartment, tissue or other compartment specified in the ICRP 30 models, including those compartments specified in the four segments of the G.I. tract. A deposition of radioelement in any of these compartments is assumed to instantaneously mix uniformly with the content of the radioelement that is present from previous depositions. This assumption results in a uniform activity distribution within a compartment, which greatly simplifies the estimation of the penetrating radiation doses delivered to various target organs as a result of transformations in various source organs. Obviously, this may be a gross simplification of the distribution of a radioelement in the various segments of the G.I. tract, where the translocation of material probably lies somewhere between the two extreme model assumptions of instantaneous uniform mixing and slug flow. If the four segments of the G.I. tract are considered as a single pipe for the slug flow model or as a vessel for the instantaneous uniform mixing model and if the same characteristic time applies (i.e., the transit time is set equal to the mean residence time), then the contents of the G.I. tract calculated by these two extreme case models would differ by only about 23%, regardless of the radioactive half-life of the radionuclide. Even though the dose distributions could differ considerably, the average doses would not be expected to differ greatly.

(4) Assumption of simple linear first order kinetics. In most cases, ICRP describes the metabolism and retention of an element within various compartments by linear first order kinetics. Biological removal from a compartment is often described by a single first order rate constant that is deemed to be applicable over the 50-year integrating period used to calculate the committed dose. In other words, within a given compartment, each element is treated as a single metabolic species whose retention in various compartments in the body is described by a particular set of biological removal rate constants over the 50-year period used to calculate the committed dose to reference man. This simplifying assumption may be reasonable, lacking more detailed information, for estimating the committed doses to workers.

(5) Assumption that daughter atoms born in the body from the decay of a radioactive parent are metabolized in the same way as parent. This assumption might be appropriate for radioactive daughter atoms born within insoluble particles that are deposited and then translocated between various compartments of the respiratory tract. It does not seem to be a reasonable assumption, however, for soluble compounds or for daughter atoms born within systemic compartments, where they would normally be born as single atoms or ions as a result of the recoil effect associated, to a lesser or greater extent, with all of the radioactive decay processes. The difficulties

with this assumption are further compounded by the fact that daughter atoms, taken into the body by themselves or in association with parent atoms, are assumed in the ICRP model to be metabolized in the same way as their own element. In this report, consideration is made for the different metabolic properties of the various radionuclides in a decay chain.

(6) Assumptions concerning the exposed individual. As in ICRP Publication 30, and except for the modifications noted for the ICRP metabolic models and risk estimates, both the estimated doses and the associated risks are calculated for an occupationally-exposed adult, referred to herein as reference man and derived from ICRP Publication 23 (ICRP 1975). The ICRP does not recommend using the data and the models to estimate doses to members of a population (for example, for radionuclides in the environment) with adjustment made solely on the basis of the differences in mass of organs or magnitude of intake. A limited bibliography concerning the estimation of committed dose equivalent in people of different ages is listed in the ICRP 30 Report. Further discussion concerning this problem is given by the NCRP (1984c) and the ICRP (1984).

In its statement from the 1983 Washington meeting, the ICRP concludes (ICRP 1984):

> The limitation of the effective dose equivalent for members of the public is sufficient to provide compliance over a lifetime with the limit for single

> organs, thus avoiding non-stochastic effects. Relative values for infants and adults of the committed dose equivalent in a number of tissues per unit intake for each of a few radionuclides have been given above: the values for infants are just more than 1 to 100 times greater than those for adult workers. In each of these cases the appropriate annual dose equivalent limits recommended by the Commission for members of the public are ten times less than the corresponding values for workers: the resulting ALI for infants aged 6 months will be smaller than the values given in ICRP Publication 30 for limiting stochastic effects in workers by factors that range from just more than 10 (for caesium-137) to 1,000 (for ingested plutonium-239). Intermediate factors would apply to older members of the public. The magnitude of the range emphasizes the need to consider each situation carefully...

4. Applications and Limitations of ICRP 30 Models to the Estimation of Internal Radiation Doses and Risks to Populations Exposed to Radioactivity in Drinking Water

As described in this report, the dose-risk model is employed to determine the health risks associated with radionuclides in drinking water to a cohort of 100,000 persons, all simultaneously liveborn and all going through life with the same risks of dying from competing causes. The age of the cohort extends from birth to 110 years. Despite the fact that members of the cohort receive exposures as both infants and adults, adult metabolic and risk parameters are used to estimate the doses and associated risks using a modified absolute risk model based upon data from the BEIR III report (NAS 1980). Inasmuch as members of the cohort receive most of their dose as adults, and lacking a unified, age-specific dose-risk model applicable to exposures to all radionuclides, this approach provides a reasonable estimate of the risks associated with radionuclides in drinking water.

Because the dosimetric model does, in fact, rely on anatomical, physiological and metabolic models for reference man, the radiation doses and associated risks in the cohort could be significantly different in certain cases. For example, when the cohort is at a younger age and in cases where a radionuclide might concentrate in a particular organ or tissue, the radiation dose following an acute intake by the cohort could be substantially larger than the dose calculated for reference man simply because of the smaller mass of the critical tissue or organ in the cohort. Specific examples might include the thyroid in the case of radioiodine and the bone or marrow in the case of bone seekers. Such examples warrant additional consideration.

It is to be noted, however, that the influence of age-specific factors on the lifetime risk for chronic exposure to a given contaminant level in drinking water should not be as great as that for acute exposures. Also, when a radionuclide has a long effective half-life in a particular organ or tissue, the lifetime risk due to larger doses at earlier ages is not likely to be much greater than that for Reference Man because most of the dose will be received as an adult even for acute exposures.

III. ASSESSMENT OF HEALTH RISK FROM EXPOSURE TO RADIOACTIVITY IN DRINKING WATER

A. Introduction

Pursuant to the Safe Drinking Water Act, the EPA is charged with identifying and regulating pollutants in drinking water that may endanger public health. This report describes the results of a dosimetric and health effects analysis for radionuclides potentially present in drinking water. The purpose of this analysis is to provide input to EPA for the development of the Revised Primary Drinking Water Regulations.

A synthesis of contemporary dosimetric and risk assessment methods has been utilized to compute estimates of risk from ingestion of radioactive pollutants in drinking water. The RADRISK computer code (Dunning et al. 1980) has been developed to estimate dose rates and subsequent health effects resulting from ingestion of a radionuclide. Dosimetry calculations are adapted from the INREM II computer code (Killough et al. 1978), and the resultant estimates of dose rate are coupled with an adaptation of the CAIRD computer code (Cook et al. 1978, Bunger et al. 1981) developed by EPA for estimation of potential health risks to a hypothetical cohort. Health effects are measured in terms of the number of incremental fatalities in the cohort, the years of life lost per premature death and the

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16. ABSTRACT

The Office of Drinking Water (ODW), U.S. Environmental Protection Agency has prepared a Drinking Water Criteria Document on MMRO.* This Criteria Document is an extensive review of the following topics:

- Physical and chemical properties of Man-made Radionuclide Occurrence
- Toxicokinetics and human exposure to MMRO*
- Health Effects of MMRO* in humans and animals
- Mechanisms of toxicity of MMRO*
- Quantification of toxicological effects of MMRO*

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
*(MMRO) properties *(MMRO) exposure *(MMRO) toxicity Drinking Water		

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EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE reduction in the cohort's life expectancy attributed to the radiation exposure.

In this analysis, the population at risk consists of a hypothetical cohort of 100,000 persons, all simultaneously liveborn and all subject to the same competing risks of death (risks other than those from man-made radiation exposure, but including risk from natural radiation) throughout their lifetimes. The probability of dying at a particular age from a competing risk is calculated from the mortality rate at that age for all causes, as specified in a recent actuarial life table for the total U.S. population (USDHEW 1975).

Both chronic and acute radiation exposures are considered in this analysis. In the chronic exposure scenario, each member of the cohort is assumed to ingest water with a unit concentration of a given radionuclide throughout his lifetime. In the acute exposure scenario, each member of the cohort is assumed to ingest a unit activity of a given radionuclide at the onset of the analysis period, with no further intake.

Estimates of incremental cancer mortality in the cohort have been computed for these chronic and acute exposures to more than 200man-made radionuclides potentially present in drinking water.

B. Estimation of Dose Rates

The RADRISK methodology has been described in detail in previous publications (Dunning et al. 1980, Sullivan et al. 1981). Only a brief summary is provided here for the reader's convenience.

The RADRISK code implements contemporary dosimetric models to estimate dose rates at various times to specified reference organs in the body from ingested radionuclides.
The dosimetric methods in RADRISK are adapted from those of the INREM II code (Killough et al. 1978), based primarily on models recommended by ICRP (1979).

In all calculations for this report, the parent nuclide of a possible chain of radioactive species (referred to as daughter products) is taken into the body by ingestion. It is assumed that no daughter radionuclides are directly ingested, so intake of daughter radionuclides must be treated in a separate execution of the code. The ingrowth and dynamics of daughter products after intake of the parent into the body, however, are considered explicitly in the calculation of dose rate. In addition, consideration is made for the different metabolic properties of the various radionuclides in the decay chain.

The change in activity within the body with time is modeled by a system of ordinary differential equations, with each equation describing the time of change of activity in a conceptual compartment of the body. Formation of radioactive daughters may occur in each compartment; each daughter may have chemical and physical properties that differ from those of the parent.

The pathways by which activity is assumed to move in the body are illustrated in Figure III-1. Movement of activity through the G.I. tract is simulated with a catenary model consisting of four segments: stomach, small intestine, upper large intestine and lower large intestine. Exponential outflow of activity from each segment into the next or out of the system is assumed. Outflow rate constants are calculated from the transit times of Eve (1966). While RADRISK permits consideration of absorption from any combination of the four segments, only activity absorbed into the blood from the small intestine is considered in this report, consistent with ICRP recommendations; the fractional absorption from the small intestine into the blood is traditionally denoted as f_1.

Activity absorbed by the blood from the G.I. tract is assumed to be distributed immediately to systemic organs. The distribution of activity to these organs is specified by fractional uptake coefficients. The list of organs in which activity is explicitly distributed (source organs) is element-dependent

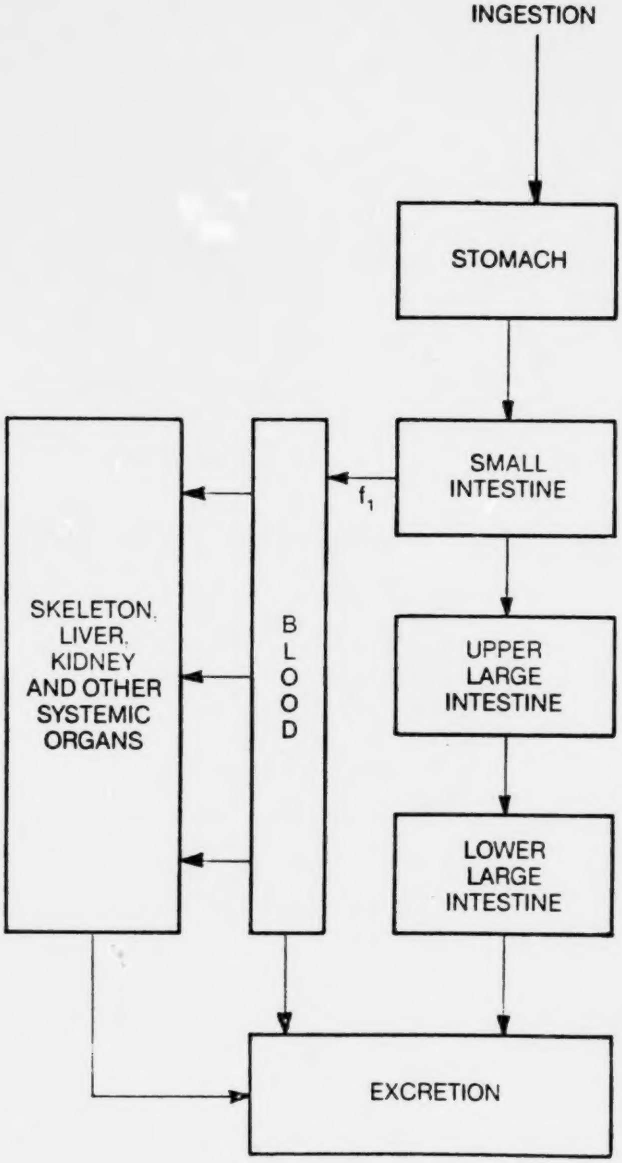
INGESTION
STOMACH
SMALL INTESTINE
f_1
UPPER LARGE INTESTINE
LOWER LARGE INTESTINE
BLOOD
SKELETON, LIVER, KIDNEY AND OTHER SYSTEMIC ORGANS
EXCRETION

Figure III-1. Schematic representation of radioactivity movement modeled by the RADRISK Code.

and may include such organs as bone or liver where appropriate metabolic data are available. This list is complemented by an additional source region, denoted as "OTHER", which accounts for that systemic activity which is not distributed among explicit source organs; uniform distribution of the activity within "OTHER" is assumed.

Radioactive material which enters an organ may be removed by both radioactive decay and biological processes. For each source organ, the fraction of the initial activity remaining at any time after intake is described by a retention function consisting of one or more exponentially decaying terms. In the present study, all retention functions are those recommended in ICRP Publication 30 (ICRP 1979, 1980, 1981).

C. Modifications of RADRISK for Acute Exposures

In this study, estimates of health risks incurred by the cohort have been computed for both chronic and acute exposures. The RADRISK code was originally designed to consider only chronic exposures, and some modifications were necessary to permit the calculation of dose rates and resultant health effects from acute radionuclide intakes.

Following an acute intake, the activity in the body decreases monotonically, particularly for radionuclides with rapid radiological decay rates or rapid biological clearance.

In the case of chronic exposure, the activity in each organ of the body increases monotonically until a steady state is achieved, at which time the activity remains constant. The instantaneous dose rates at various times after the start of chronic exposure provide a reasonably accurate (and conservative) estimate of the total annual dose for chronic exposure conditions. The instantaneous dose rates may, however, err substantially in the estimation of annual dose from an acute exposure, particularly if the activity levels decrease rapidly.

For this reason, the RADRISK code was modified to compute the total dose in each year following exposure and to use these values in the estimation of incremental cancer mortality in the cohort. Since the rate of change in activity levels in various organs is more rapid at early times after exposure, doses are computed annually for the first several years and for progressively longer periods thereafter, dividing by the length of the interval to estimate the average annual dose. This method produces estimates of risk that are similar to those computed by the original RADRISK methodology for chronic exposures and provides a more accurate estimate of the risks from acute intakes.

For both chronic and acute exposures, it is assumed that each member of the cohort is exposed to a unit activity of a given radionuclide. The cohort consists of an initial population of 100,000 persons, all of whom are simultaneously newborn and subject to the same competing risks. In the chronic exposure scenario, the radiation exposure is assumed to begin at birth and continue throughout the entire lifetime of each individual. The average lifetime of the cohort is 70.7 years, and no member of the cohort lives more than 110 years. The span from 0 to 110 years is divided into nine age intervals (0 to 2, 2 to 4, 4 to 8, 8 to 16, 16 to 24, 24 to 36, 36 to 48, 48 to 64 and 64 to 110), and dose rates at the midpoints of the age intervals (i.e., at years 1, 3, 6, 12, 20, 30, 42, 56 and 87, respectively) are used as estimates of the annual doses during those intervals.

In the acute exposure scenario, each member of the cohort ingests 1 pCi (0.037 Bq) of a given radionuclide in a single day. This exposure occurs at age 20, with no radiation exposure thereafter. The period between 20 and 110 years is divided into nine age intervals (20 to 21, 21 to 22, 22 to 23, 23 to 25, 25 to 30, 30 to 40, 40 to 50, 50 to 70 and 70 to 110), and RADRISK computes the accumulated dose in each of these subintervals, dividing by the length of the subinterval to estimate the average annual dose.

D. Estimation of Incremental Risk

In the present study, the risk from radionuclide exposures is expressed in terms of incremental cancer fatalities. Genetic effects and non-fatal somatic effects are not considered.

The incremental cancer mortality is estimated in the following manner.

For each organ, instantaneous dose rates at specified times (for chronic exposures) or annual doses (for acute exposures) are processed through a subroutine adaptation of the CAIRD code (Cook et al. 1978). This subroutine uses the annual doses (or those derived from the transmitted dose rates) to estimate the number of incremental fatalities in the cohort due to the additional radiation-induced cancers in the reference organ. The calculation of incremental fatalities is based on estimated annual incremental risks, which are computed from annual doses to the organ, together with radiation risk factors, such as those given in the Committee on the Biological Effects of Ionizing Radiation reports (BEIR) (NAS 1972, 1980).

An important feature of this methodology is the use of an actuarial life table to account for the time dependence of the radiation insult and to allow for competing risks of death in the estimation of risk due to radiation exposure. A life table consists of data describing age-specific mortality rates from all causes of death for a given population. This information is derived from data obtained on actual mortality rates in a real population.

The use of life tables in studies of risk due to low-level radiation exposure is important because of the time delay inherent in radiation risk. After a radiation dose is received, there is a latent (or induction) period of some years before a cancer may appear. Following this latency period, the probability of a cancer occurring during a given year is assumed to be constant for a specified period of time, called the plateau period, after which the risk is assumed to vanish. The lengths of the latency and plateau periods depend upon the type of cancer and upon the age and other characteristics of the persons at risk.

During or after radiation exposure, a potential cancer victim may experience years of life in which he is continually exposed to risks of death from causes other than radiation exposure. Hence, some individuals will be lost from the population due to competing causes of death and are not potential victims of radiation-induced cancer. If the lengths of the latent and plateau periods are known, the life tables can be used to estimate the number of individuals that will die from additional radiation-induced cancers.

For a given organ, the incremental probability of death due to radiation-induced cancer is estimated for each year using radiation risk factors and the estimated doses during that year and relevant preceding years. The incremental probabilities of death are used in conjunction with actuarial life tables to estimate the incremental number of radiation-induced deaths each year. The life tables used throughout this analysis are based on statistics for the total U.S. population during the years 1969 to 1971 (USDHEW 1975).

In addition to providing an estimate of the incremental number of deaths, the life table methodology can be used to estimate the total number of years of life lost to those dying of radiation-induced cancer, the average number of years of life lost per incremental mortality and the decrease in the life expectancy of the population. The total number of years of life lost to those dying of radiation-induced cancer is computed as the difference between the total number of years of life lived by the cohort, assuming no incremental radiation risk, and the total number of years of life lived by the same cohort, assuming the incremental risk from radiation. The decrease in the life expectancy of the population can be calculated as the total years of life lost divided by the original cohort size.

Either absolute or relative risk factors can be used in the RADRISK code. Absolute risk factors, given in terms of deaths per unit dose, are based on the assumption that there is some absolute number of deaths in a population per unit of dose. Relative risk factors, given in terms of a percentage increase in cancer deaths per unit dose, are based on the assumption that the number of radiation-induced cancer deaths associated with a given type of cancer depends on the total number of naturally-occurring fatal cancers of that type.

Both absolute and relative risk factors are assumed to apply uniformly during a plateau period, beginning at the end of a latent period. Different risk factors are often used for adult, child and _in utero_ exposures. In this analysis, however, only adult risk parameters and a modified absolute risk model are considered.

E. Assessment of Potential Risks

The methodology described in the preceding sections has been used to compute estimates of potential health risks from ingestion of various radionuclides in drinking water. Only the risk of fatal cancers in each of the 18 body organs shown in Table III-1 is considered in these estimates. Risks of non-fatal cancers, genetic risks and risks from _in utero_ exposures are not considered in this analysis.

Table III-1 gives the latency and plateau periods which are assumed for each organ at risk, as well as risk factors for high-LET (principally alpha radiation) and low-LET radiation. Application of the absolute and relative risk factors from the 1972 BEIR report (NAS 1972), with assumptions of 30-year and lifetime plateau periods for expression of solid tumors and a 25-year plateau for leukemia, to a cohort with the mortality characteristics of the 1969 to 1971 U.S. population produced estimates of 85 to 470 incremental deaths in the cohort from a chronic dose rate of 0.1 rem/year. These estimates were averaged to obtain an estimated incremental risk to the cohort of 200 excess cancer deaths from this chronic 0.1-rem/year dose rate. The organ-specific risk estimates in Table III-1 were obtained by distributing the estimated 200 excess cancer mortalities among the selected organs in accordance with the best available data (NAS 1972, 1980, UNSCEAR 1972, 1977).

It is assumed that the probability of occurrence for each type of cancer is directly proportional to the radiation dose received, without threshold. This assumption that the cancer probability is linear (commonly referred to as the "linear hypothesis") is used throughout these calculations. The validity of the linear hypothesis is a subject of continuing controversy (NAS 1980). Selection of the dose-response model to be used for extrapolations affects predictions enormously, and the linear hypothesis represents only one of the several models which have been proposed. Its use in this study is based on consideration of available information and limitations of computational models; the linear model has the attractive feature of allowing results based upon a unit exposure to be scaled to reflect a specific exposure situation.

For chronic exposures, it was assumed that the cohort of 100,000 persons was exposed to a constant intake of 1 pCi/year (0.037 Bq/year) of a radionuclide for the lifetime of the cohort. Dose rates and risk parameters for a reference adult were assumed to apply to all members of the cohort at all

Table III-1 Risk Parameters for Cancers Considered

Cancer	Latency (years)	Plateau (years)	Risk factor for low-LET radiation (deaths/10^6 rad person years at risk)	Risk factor for high-LET raditation (deaths/10^6 rad person years at risk)	Number of premature deaths in cohort from chronic 1-mrad/year exposure[a]
Leukemia - red marrow	2	25	2.3	46	0.326
Bone surface	5	30	0.2	2	0.031
Lung	10	110	3.0	30	0.608
Breast	15	110	2.3	2.3	0.399
Liver	15	110	0.9	9	0.156
Stomach	15	110	0.5	5	0.087
Pancreas	15	110	0.7	7	0.121
Lower large intestine	15	110	0.4	4	0.069
Kidneys	15	110	0.2	2	0.035
Bladder	15	110	0.2	2	0.035

---

[a] Low LET.

- continued

Table III-1 - continued

Cancer	Latency (years)	Plateau (years)	Risk factor for low-LET radiation (death 10^6 rad at risk)	Risk factor for high-LET radiation (death 10^6 rad at risk)	Number of premature deaths in cohort from chronic 1-mrad/year exposure
Upper large intestine	15	110	0.2	2	0.035
Small intestine	15	110	0.1	1	0.017
Ovaries	15	110	0.1	1	0.017
Testes	15	110	0.1	1	0.017
Spleen	15	110	0.1	1	0.017
Uterus	15	110	0.1	1	0.017
Thymus	15	110	0.1	1	0.017
Thyroid	2	45	0.4[b]	0.4[b]	0.085

[a] Low LET.

[b] For ^{131}I and longer-lived radioiodine, 0.04.

times. Since the models used in RADRISK are linear, these results may be scaled to evaluate any other chronic exposure conditions.

Estimates of risk resulting from chronic exposure to each of more than 200 radionuclides are presented in Table III-2. Results are presented as the number of excess fatal radiation-induced cancers in the cohort produced by the unit exposure. For some elements, absorption from the G.I. tract, characterized here by the parameter, f_1, may vary significantly depending on the chemical form. Table III-2 lists multiple cases for isotopes of sulfur, chromium, cobalt, selenium, strontium, molybdenum, antimony, tungsten and uranium. The different chemical compounds represented by each f_1 value are indicated in Table III-3.

Throughout these calculations, metabolic parameters and specific effective energies (SEEs) recommended by ICRP (1979, 1980, 1981) have been employed. One important systematic difference between the calculations performed here and those in ICRP Publication 30 is the treatment of radioactive daughters produced in the body. The ICRP assumes that all radioactive progeny produced in the body follow the metabolic behavior of the parent. In the calculations performed here, however, each daughter is permitted to assume the metabolic characteristics appropriate for that element. This assumption can impact estimates of dose and subsequent risk significantly.

Table III-2 Premature Deaths from Chronic Radionuclide Ingestion and Derived Radionuclide Concentrations in Drinking Water Producing Risk from Chronic Ingestion of 2 L/day Equal to That from a Constant Dose Rate of 4-mrem/year

Radio-nuclide	f_1	Incremental deaths in cohort of 10^5 persons from ingestion of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
H-3	1.0E-00	1.E-07	9.E 04
Be-7	5.0E-03	9.E-08	1.E 05
C-14	1.0E-00	4.E-06	3.E 03
F-18	1.0E-00	2.E-07	5.E 04
Na-22	1.0E-00	2.E-05	5.E 02
Na-24	1.0E-00	3.E-06	4.E 03
Si-31	1.0E-02	4.E-07	3.E 04
P-32	8.0E-01	2.E-05	7.E 02
S-35	8.0E-01	8.E-07	1.E 04
	1.0E-01	8.E-07	1.E 04
Cl-36	1.0E-00	6.E-06	2.E 03
Cl-38	1.0E-00	4.E-07	3.E 04
K-40	1.0E-00	4.E-05	3.E 02
K-42	1.0E-00	2.E-06	5.E 03
Ca-45	3.0E-01	6.E-06	2.E 03
Ca-47	3.0E-01	8.E-06	1.E 03
Sc-46	1.0E-04	5.E-06	2.E 03
Sc-47	1.0E-04	2.E-06	5.E 03
Sc-48	1.0E-04	6.E-06	2.E 03
V-48	1.0E-02	7.E-06	2.E 03
Cr-51	1.0E-01	1.E-07	8.E 04
Cr-51	1.0E-02	1.E-07	9.E 04
Mn-52	1.0E-01	7.E-06	2.E 03
Mn-54	1.0E-01	4.E-06	3.E 03
Mn-56	1.0E-01	8.E-07	1.E 04
Fe-55	1.0E-01	1.E-06	1.E 04
Fe-59	1.0E-01	9.E-06	1.E 03
Co-57	3.0E-01	2.E-06	6.E 03
	5.0E-02	8.E-07	1.E 04
Co-58m	3.0E-01	9.E-08	1.E 05
	5.0E-02	8.E-08	1.E 05
Co-58	3.0E-01	5.E-06	2.E 03
	5.0E-02	3.E-06	4.E 03
Co-60	3.0E-01	5.E-05	2.E 02
	5.0E-02	1.E-05	9.E 02

- continued

Table III-2 - continued

Radio-nuclide	f_1	Incremental deaths in cohort of 10^5 persons from ingestions of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
Ni-59	5.0E-02	3.E-07	3.E 04
Ni-63	5.0E-02	8.E-07	1.E 04
Ni-65	5.0E-02	5.E-07	2.E 04
Cu-64	5.0E-01	5.E-07	2.E 04
Zn-65	5.0E-01	3.E-05	4.E 02
Zn-69m	5.0E-01	1.E-06	8.E 03
Zn-69	5.0E-01	9.E-08	1.E 05
Ga-72	1.0E-03	4.E-06	3.E 03
Ge-71	1.0E-00	2.E-08	7.E 05
As-73	5.0E-01	9.E-07	1.E 04
As-74	5.0E-01	5.E-06	2.E 03
As-76	5.0E-01	5.E-06	2.E 03
As-77	5.0E-01	1.E-06	8.E 03
Se-75	8.0E-01	2.E-05	6.E 02
	5.0E-02	2.E-06	5.E 03
Br-82	1.0E-00	3.E-06	3.E 03
Rb-86	1.0E-00	2.E-05	6.E 02
Rb-87	1.0E-00	2.E-05	6.E 02
Sr-85m	3.0E-01	2.E-08	4.E 05
	1.0E-02	2.E-08	5.E 05
Sr-85	3.0E-01	3.E-06	4.E 03
	1.0E-02	1.E-06	1.E 04
Sr-89	3.0E-01	1.E-05	9.E 02
	1.0E-02	9.E-06	1.E 03
Sr-90	3.0E-01	2.E-04	5.E 01
	1.0E-02	2.E-05	7.E 02
Sr-91	3.0E-01	2.E-06	5.E 03
	1.0E-02	3.E-06	4.E 03
Sr-92	3.0E-01	1.E-06	8.E 03
	1.0E-02	2.E-06	7.E 03
Y-90	1.0E-04	1.E-05	1.E 03
Y-91m	1.0E-04	4.E-08	2.E 05
Y-91	1.0E-04	9.E-06	1.E 03
Y-92	1.0E-04	1.E-06	7.E 03
Y-93	1.0E-04	4.E-06	3.E 03
Zr-93	2.0E-03	1.E-06	1.E 04
Zr-95	2.0E-03	3.E-06	3.E 03
Zr-97	2.0E-03	7.E-06	2.E 03
Nb-93m	1.0E-02	5.E-07	2.E 04
Nb-95m	1.0E-02	2.E-06	5.E 03
Nb-95	1.0E-02	2.E-06	5.E 03
Nb-97m	1.0E-02	4.E-09	3.E 06

-continued

Table III-2 - continued

Radionuclide	f_1	Incremental deaths in cohort of 10^5 persons from ingestion of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
Nb-97	1.0E-02	2.E-07	5.E 04
Mo-99	8.0E-01	5.E-06	2.E 03
Mo-99	5.0E-02	5.E-06	2.E 03
Tc-96m	8.0E-01	5.E-08	2.E 05
Tc-96	8.0E-01	4.E-06	3.E 03
Tc-97m	8.0E-01	2.E-06	5.E 03
Tc-97	8.0E-01	3.E-07	4.E 04
Tc-99m	8.0E-01	9.E-08	1.E 05
Tc-99	8.0E-01	2.E-06	5.E 03
Ru-97	5.0E-02	5.E-07	2.E 04
Ru-103	5.0E-02	3.E-06	4.E 03
Ru-106	5.0E-02	3.E-05	3.E 02
Rh-103m	5.0E-02	1.E-08	1.E 06
Rh-105	5.0E-02	1.E-06	8.E 03
Rh-106	5.0E-02	6.E-09	2.E 06
Pd-103	5.0E-03	7.E-07	1.E 04
Pd-109	5.0E-03	2.E-06	5.E 03
Ag-105	5.0E-02	2.E-06	5.E 03
Ag-109m	5.0E-02	5.E-10	2.E 07
Ag-110m	5.0E-02	1.E-05	7.E 02
Ag-110	5.0E-02	4.E-09	3.E 06
Ag-111	5.0E-02	5.E-06	2.E 03
Cd-109	5.0E-02	2.E-05	5.E 02
Cd-115m	5.0E-02	2.E-05	7.E 02
Cd-115	5.0E-02	5.E-06	2.E 03
In-113m	2.0E-02	9.E-08	1.E 05
In-114m	2.0E-02	2.E-05	5.E 02
In-114	2.0E-02	8.E-09	1.E 06
In-115m	2.0E-02	3.E-07	4.E 04
In-115	2.0E-02	2.E-04	6.E 01
Sn-113	2.0E-02	3.E-06	4.E 03
Sn-125	2.0E-02	1.E-05	9.E 02
Sb-122	1.0E-01	7.E-06	2.E 03
	1.0E-02	7.E-06	2.E 03
Sb-124	1.0E-01	1.E-05	1.E 03
	1.0E-02	9.E-06	1.E 03
Sb-125	1.0E-01	3.E-06	4.E 03
	1.0E-02	2.E-06	4.E 03
Te-125m	2.0E-01	5.E-06	2.E 03
Te-127m	2.0E-01	1.E-05	9.E 02
Te-127	2.0E-01	6.E-07	2.E 04
Te-129m	2.0E-01	1.E-05	8.E 02

- continued

Table III-2 - continued

Radio-nuclide	f_1	Incremental deaths in cohort of 10^5 persons from ingestion of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
Te-129	2.0E-01	2.E-07	6.E 04
Te-131m	2.0E-01	2.E-05	7.E 02
Te-131	2.0E-01	4.E-07	2.E 04
Te-132	2.0E-01	2.E-05	6.E 02
I-125	1.0E-00	1.E-05	1.E 03
I-126	1.0E-00	2.E-05	5.E 02
I-129	1.0E-00	8.E-05	1.E 02
I-131	1.0E-00	2.E-05	7.E 02
I-132	1.0E-00	2.E-06	7.E 03
I-133	1.0E-00	3.E-05	4.E 02
I-134	1.0E-00	5.E-07	2.E 04
I-135	1.0E-00	6.E-06	2.E 03
Cs-131	1.0E-00	5.E-07	2.E 04
Cs-134m	1.0E-00	9.E-08	1.E 05
Cs-134	1.0E-00	1.E-04	8.E 01
Cs-135	1.0E-00	1.E-05	8.E 02
Cs-136	1.0E-00	2.E-05	5.E 02
Cs-137	1.0E-00	1.E-04	1.E 02
Ba-131	1.0E-01	2.E-06	7.E 03
Ba-137m	1.0E-01	4.E-09	3.E 06
Ba-140	1.0E-01	1.E-05	1.E 03
La-140	1.0E-03	7.E-06	1.E 03
Ce-141	3.0E-04	3.E-06	4.E 03
Ce-143	3.0E-04	4.E-06	3.E 03
Pr-142	3.0E-04	5.E-06	2.E 03
Pr-143	3.0E-04	5.E-06	2.E 03
Nd-149	3.0E-04	4.E-07	3.E 04
Pm-149	3.0E-04	4.E-06	3.E 03
Sm-151	3.0E-04	4.E-07	3.E 04
Sm-153	3.0E-04	3.E-06	4.E 03
Eu-152	1.0E-03	7.E-06	2.E 03
Eu-154	1.0E-03	1.E-05	1.E 03
Eu-155	1.0E-03	2.E-06	7.E 03
Gd-153	3.0E-04	1.E-06	1.E 04
Gd-159	3.0E-04	2.E-06	6.E 03
Tb-160	3.0E-04	6.E-06	2.E 03
Dy-165	3.0E-04	3.E-07	4.E 04
Dy-166	3.0E-04	7.E-06	2.E 03
Ho-166	3.0E-04	5.E-06	2.E 03
Er-169	3.0E-04	1.E-06	7.E 03
Er-171	3.0E-04	1.E-06	9.E 03
Tm-170	3.0E-04	5.E-06	2.E 03

- continued

Table III-2 (continued)

Radio-nuclide[a]	f_1	Incremental deaths in cohort of 10^5 persons from ingestion of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
Tm-171	3.0E-04	4.E-07	3.E 04
Yb-175	3.0E-04	2.E-06	7.E 03
Lu-177	3.0E-04	2.E-06	5.E 03
Hf-181	2.0E-03	4.E-06	3.E 03
Ta-182	1.0E-03	6.E-06	2.E 03
W-181	3.0E-01	3.E-07	4.E 04
	1.0E-02	3.E-07	4.E 04
W-185	3.0E-01	2.E-06	7.E 03
	1.0E-02	2.E-06	6.E 03
W-187	3.0E-01	2.E-06	6.E 03
	1.0E-02	2.E-06	5.E 03
Re-183	8.0E-01	2.E-06	7.E 03
Re-186	8.0E-01	5.E-06	2.E 03
Re-187	8.0E-01	2.E-08	7.E 05
Re-188	8.0E-01	5.E-06	2.E 03
Os-185	1.0E-02	2.E-06	5.E 03
Os-191m	1.0E-02	3.E-07	3.E 04
Os-191	1.0E-02	2.E-06	5.E 03
Os-193	1.0E-02	3.E-06	4.E 03
Ir-190	1.0E-02	5.E-06	2.E 03
Ir-192	1.0E-02	6.E-06	2.E 03
Ir-194	1.0E-02	5.E-06	2.E 03
Pt-191	1.0E-02	1.E-06	9.E 03
Pt-193m	1.0E-02	2.E-06	6.E 03
Pt-193	1.0E-02	1.E-07	9.E 04
Pt-197m	1.0E-02	3.E-07	4.E 04
Pt-197	1.0E-02	1.E-06	8.E 03
Au-196	1.0E-01	1.E-06	9.E 03
Au-198	1.0E-01	4.E-06	3.E 03
Hg-197	2.0E-02	9.E-07	1.E 04
Hg-203	2.0E-02	2.E-06	5.E 03
Tl-202	1.0E-00	3.E-06	4.E 03
Tl-204	1.0E-00	6.E-06	2.E 03
*Tl-207	1.0E-00	2.E-08	NA
*Tl-208	1.0E-00	3.E-08	NA
Pb-203	2.0E-01	1.E-06	1.E 04
*Pb-210	2.0E-01	3.E-04	NA
*Pb-211	2.0E-01	4.E-07	NA
*Pb-212	2.0E-01	4.E-05	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

Table III-2 - continued

Radio-nuclide[a]	f_1	Incremental deaths in cohort of 10^5 persons from ingestion of 1.0 pCi/year	Concentration (pCi/L) in drinking water yielding a risk equal to that from a dose rate of 4 mrem/year
*Pb-214	2.0E-01	4.E-07	NA - 3,000
Bi-206	5.0E-02	7.E-06	2.E 03
Bi-207	5.0E-02	5.E-06	2.E 03
*Bi-210	5.0E-02	9.E-06	NA
*Bi-211	5.0E-02	2.E-08	NA
*Bi-212	5.0E-02	6.E-07	NA
*Bi-214	5.0E-02	2.E-07	NA
*Po-210	1.0E-01	7.E-04	NA
*Po-212	1.0E-01	4.E-17	NA
*Po-214	1.0E-01	2.E-14	NA
*Po-215	1.0E-01	5.E-13	NA
*Po-216	1.0E-01	2.E-10	NA
*Po-218	1.0E-01	6.E-08	NA
*Ra-223	2.0E-01	7.E-04	NA
*Ra-224	2.0E-01	4.E-04	NA
*Ra-226	2.0E-01	1.E-03	NA
*Ra-228	2.0E-01	6.E-04	NA
*Ac-227	1.0E-03	3.E-03	NA
*Ac-228	1.0E-02	1.E-06	NA
*Th-227	2.0E-04	3.E-05	NA
*Th-228	2.0E-04	2.E-04	NA
*Th-230	2.0E-04	4.E-04	NA
*Th-231	2.0E-04	1.E-06	NA
*Th-232	2.0E-04	3.E-04	NA
*Th-234	2.0E-04	1.E-05	NA
*Pa-231	1.0E-03	3.E-03	NA
Pa-233	1.0E-03	3.E-06	3.E 03
*Pa-234m	1.0E-03	8.E-09	NA
*Pa-234	1.0E-03	2.E-06	NA
*U-234	2.0E-01	6.E-04	NA
	5.0E-02	2.E-04	NA
	2.0E-03	1.E-05	NA
*U-235	2.0E-01	6.E-04	NA
	5.0E-02	1.E-04	NA
	2.0E-03	2.E-05	NA
*U-238	2.0E-01	6.E-04	NA
	5.0E-02	1.E-04	NA
	2.0E-03	1.E-05	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

Also computed for each radionuclide is the concentration in drinking water that produces a risk from chronic ingestion of 2 L/day which is approximately equivalent to the risk from a uniform whole body dose rate of 4-mrem/year (40 Sv/year). This risk is estimated to be approximately eight excess fatal cancers in the cohort. The concentration of each radionuclide that produces a risk equal to that resulting from a chronic 4-mrem/year dose rate is computed as:

Reference Radionuclide Concentration (pCi/L) =

$$\frac{\text{8 excess deaths}}{\begin{array}{c}\text{lifetime cohort exposure}\\ \text{to 4-mrem/year dose rate}\end{array}} \times \left[\frac{\text{number of excess deaths}}{\begin{array}{c}\text{1-pCi/year lifetime cohort}\\ \text{exposure to the nuclide}\end{array}}\right]^{-1}$$

$$\times \left[\frac{730\ \text{L}}{\text{year}}\right]^{-1} = 1.1 \times 10^{-2}$$

$$\times \left[\frac{\text{number of excess deaths in cohort}}{\begin{array}{c}\text{1-pCi/year lifetime cohort exposure}\\ \text{to the radionuclide}\end{array}}\right]^{-1}$$

This calculation assumes an average lifetime of 70.7 years for members of the cohort and a risk of 200 excess fatal cancers in the cohort from a chronic uniform whole body dose rate of 0.1 rem/year (computed as the geometric mean of risk estimates, based on the absolute and relative risk models). These reference concentrations are shown in the last column of Table III-2.

Table III-3 Elements with Multiple f_1 Values[a]

Element	f_1	Compound
Sulfur (S)	0.1	Elemental sulfur.
	0.8	All inorganic compounds.
Chromium (Cr)	0.01	Trivalent chromium compounds.
	0.1	Hexavalent chromium compounds.
Cobalt (Co)	0.05	Oxides, hydroxides and all other inorganic compounds.
	0.3	Organically complexed compounds and inorganic compounds, except oxides and hydroxides, in the presence of carrier material.
Selenium (Se)	0.05	Elemental selenium and selenides.
	0.8	All other compounds.
Strontium (Sr)	0.01	$SrTiO_3$.
	0.3	Soluble salts.
Molybdenum (Mo)	0.05	MoS_2.
	0.8	All compounds other than MoS_2.
Antimony (Sb)	0.01	All compounds other than tartar emetic.
	0.1	Tartar emetic.
Tungsten (W)	0.01	Tungstic acid.
	0.3	All compounds other than tungstic acid.

[a] Classifications from ICRP Publication 30 (ICRP 1979, 1980, 1981).

The numbers of excess fatal cancers experienced by the cohort following an acute ingestion of 1 pCi (0.037 Bq) of each radionuclide are presented in Table III-4. The radionuclide ingestion is assumed to occur at age 20 and to last for only a single day. Again, dose rates and risk parameters for a reference adult are assumed to apply to all members of the cohort at all times.

Tables III-4 to III-6 also present the concentrations of each radionuclide in drinking water that produce a risk equal to that from hypothetical uniform whole body doses of 50, 170, 500 and 5,000 mrem. These concentrations are based on ingestion of 2 L/day for periods of one day, ten days and one year (Tables III-4, III-5 and III-6, respectively). Since a linear dose-response model is assumed throughout this analysis, concentrations of any radionuclide yielding a risk equal to that from any other reference dose may be readily computed by linear scaling. Similarly, other exposure durations may be considered by dividing the concentration estimates in Table III-4 (one-day exposure duration) by the desired exposure period (in days). Tables III-4 and III-5, as well as the multiple reference doses in Tables III-4 to III-6, are provided only for the reader's convenience.

For each radionuclide, the reference concentration in drinking water is computed as the ratio of the risk (number of premature deaths in the cohort) from the reference dose

Table III-4 Premature Deaths from Acute Radionuclide Ingestion and Derived Radionuclide Concentrations in Drinking Water Producing Risk from Ingestion of 2 L/day for One Day Equal to That from Reference Radiation Doses

Radionuclide	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
H-3	1.0E-00	2.E-09	4.E 08	1.E 09	4.E 09	4.E 10
Be-7	5.0E-03	2.E-09	5.E 08	2.E 09	5.E 09	5.E 10
C-14	1.0E-00	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
F-18	1.0E-00	4.E-09	2.E 08	8.E 08	2.E 09	2.E 10
Na-22	1.0E-00	4.E-07	2.E 06	8.E 06	2.E 07	2.E 08
Na-24	1.0E-00	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Si-31	1.0E-02	9.E-09	1.E 08	4.E 08	1.E 09	1.E 10
P-32	8.0E-01	3.E-07	3.E 06	1.E 07	3.E 07	3.E 08
S-35	8.0E-01	2.E-08	7.E 07	2.E 08	7.E 08	7.E 09
	1.0E-01	2.E-08	7.E 07	2.E 08	7.E 08	7.E 09
Cl-36	1.0E-00	1.E-07	8.E 06	3.E 07	8.E 07	8.E 08
Cl-38	1.0E-00	7.E-09	1.E 08	5.E 08	1.E 09	1.E 10
K-40	1.0E-00	8.E-07	1.E 06	5.E 06	1.E 07	1.E 08
K-42	1.0E-00	4.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Ca-45	3.0E-01	1.E-07	1.E 07	3.E 07	1.E 08	1.E 09
Ca-47	3.0E-01	2.E-07	7.E 06	2.E 07	7.E 07	7.E 08
Sc-46	1.0E-04	1.E-07	1.E 07	3.E 07	1.E 08	1.E 09
Sc-47	1.0E-04	4.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Sc-48	1.0E-04	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
V-48	1.0E-02	1.E-07	7.E 06	2.E 07	7.E 07	7.E 08
Cr-51	1.0E-01	3.E-09	4.E 08	1.E 09	4.E 09	4.E 10
	1.0E-02	2.E-09	4.E 08	1.E 09	4.E 09	4.E 10
Mn-52	1.0E-01	1.E-07	7.E 06	3.E 07	7.E 07	7.E 08
Mn-54	1.0E-01	7.E-08	1.E 07	5.E 07	1.E 08	1.E 09
Mn-56	1.0E-01	2.E-08	6.E 07	2.E 08	6.E 08	6.E 09
Fe-55	1.0E-01	2.E-08	5.E 07	2.E 08	5.E 08	5.E 09
Fe-59	1.0E-01	2.E-07	6.E 06	2.E 07	6.E 07	6.E 08
Co-57	3.0E-01	2.E-08	6.E 07	2.E 08	6.E 08	6.E 09
	5.0E-02	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
Co-58m	3.0E-01	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
	5.0E-02	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
Co-58	3.0E-01	5.E-08	2.E 07	6.E 07	2.E 08	2.E 09
	5.0E-02	6.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Co-60	3.0E-01	9.E-07	1.E 06	4.E 06	1.E 07	1.E 08
	5.0E-02	3.E-07	4.E 06	1.E 07	4.E 07	4.E 08

- continued

Table III-4 - continued

Radio-nuclide	f_1	Incre-mental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
Ni-59	5.0E-02	6.E-09	2.E 08	5.E 08	2.E 09	2.E 10
Ni-63	5.0E-02	2.E-08	6.E 07	2.E 08	6.E 08	6.E 09
Ni-65	5.0E-02	1.E-08	1.E 08	3.E 08	1.E 09	1.E 10
Cu-64	5.0E-01	1.E-08	1.E 08	4.E 08	1.E 09	1.E 10
Zn-65	5.0E-01	5.E-07	2.E 06	6.E 06	2.E 07	2.E 08
Zn-69m	5.0E-01	3.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Zn-69	5.0E-01	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
Ga-72	1.0E-03	8.E-08	1.E 07	5.E 07	1.E 08	1.E 09
Ge-71	1.0E-00	3.E-10	3.E 09	1.E 10	3.E 10	3.E 11
As-73	5.0E-01	2.E-08	6.E 07	2.E 08	6.E 08	6.E 09
As-74	5.0E-01	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
As-76	5.0E-01	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
As-77	5.0E-01	3.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Se-75	8.0E-01	4.E-07	3.E 06	1.E 07	3.E 07	3.E 08
	5.0E-02	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
Br-82	1.0E-00	7.E-08	2.E 07	5.E 07	2.E 08	2.E 09
Rb-86	1.0E-00	4.E-07	3.E 06	9.E 07	3.E 07	3.E 08
Rb-87	1.0E-00	3.E-07	3.E 06	1.E 07	3.E 07	3.E 08
Sr-85m	3.0E-01	5.E-10	2.E 09	7.E 09	2.E 10	2.E 11
	1.0E-02	5.E-10	2.E 09	7.E 09	2.E 10	2.E 11
Sr-85	3.0E-01	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
	1.0E-02	2.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Sr-89	3.0E 01	2.E-07	4.E 06	1.E 07	4.E 07	4.E 08
	1.0E-02	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08
Sr-90	3.0E-01	4.E-06	2.E 05	8.E 05	2.E 06	2.E 07
	1.0E-02	3.E-07	3.E 06	1.E 07	3.E 07	3.E 08
Sr-91	3.0E-01	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
	1.0E-02	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Sr-92	3.0E-01	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
	1.0E-02	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Y-90	1.0E-04	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08
Y-91m	1.0E-04	9.E-10	1.E 09	4.E 09	1.E 10	1.E 11
Y-91	1.0E-04	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08
Y-92	1.0E-04	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Y-93	1.0E-04	8.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Zr-93	2.0E-03	2.E-08	5.E 07	2.E 08	5.E 08	5.E 09
Zr-95	2.0E-03	7.E-08	2.E 07	5.E 07	2.E 08	2.E 09
Zr-97	2.0E-03	2.E-07	7.E 06	2.E 07	7.E 07	7.E 08

- continued

Table III-4 - continued

Radionuclide	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
Nb-93m	1.0E-02	1.E-08	1.E 08	3.E 08	1.E 09	1.E 10
Nb-95m	1.0E-02	5.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Nb-95	1.0E-02	4.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Nb-97m	1.0E-02	9.E-11	1.E 10	4.E 10	1.E 11	1.E 12
Nb-97	1.0E-02	4.E-09	2.E 08	8.E 08	2.E 09	2.E 10
Mo-99	8.0E-01	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Mo-99	5.0E-02	1.E-07	1.E 07	3.E 07	1.E 08	1.E 09
Tc-96m	8.0E-01	1.E-09	1.E 09	4.E 09	1.E 10	1.E 11
Tc-96	8.0E-01	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Tc-97m	8.0E-01	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
Tc-97	8.0E-01	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10
Tc-99m	8.0E-01	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
Tc-99	8.0E-01	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Ru-97	5.0E-02	1.E-08	9.E 07	3.E 08	9.E 08	9.E 09
Ru-103	5.0E-02	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Ru-106	5.0E-02	7.E-07	2.E 06	5.E 06	2.E 07	2.E 08
Rh-103m	5.0E-02	2.E-10	5.E 09	2.E 10	5.E 10	5.E 11
Rh-105	5.0E-02	3.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Rh-106	5.0E-02	1.E-10	8.E 09	3.E 10	8.E 10	8.E 11
Pd-103	5.0E-03	2.E-08	7.E 07	2.E 08	7.E 08	7.E 09
Pd-109	5.0E-03	5.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Ag-105	5.0E-02	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Ag-109m	5.0E-02	1.E-11	1.E 11	4.E 11	1.E 12	1.E 13
Ag-110m	5.0E-02	3.E-07	3.E 06	1.E 07	3.E 07	3.E 08
Ag-110	5.0E-02	9.E-11	1.E 10	4.E 10	1.E 11	1.E 12
Ag-111	5.0E-02	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
Cd-109	5.0E-02	5.E-07	2.E 06	7.E 06	2.E 07	2.E 08
Cd-115m	5.0E-02	3.E-07	3.E 06	1.E 07	3.E 07	3.E 08
Cd-115	5.0E-02	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
In-113m	2.0E-02	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
In-114m	2.0E-02	4.E-07	3.E 06	9.E 06	3.E 07	3.E 08
In-114	2.0E-02	2.E-10	6.E 09	2.E 10	6.E 10	6.E 11
In-115m	2.0E-02	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10
In-115	2.0E-02	4.E-06	3.E 05	1.E 06	3.E 06	3.E 07
Sn-113	2.0E-02	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Sn-125	2.0E-02	3.E-07	4.E 06	1.E 07	4.E 07	4.E 08
Sb-122	1.0E-01	1.E-07	8.E 06	3.E 07	8.E 07	8.E 08
Sb-122	1.0E-02	1.E-07	7.E 06	2.E 07	7.E 07	7.E 08
Sb-124	1.0E-01	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08

- continued

Table III-4 - continued

Radionuclide	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
Sb-124	1.0E-02	2.E-07	6.E 06	2.E 07	6.E 07	6.E 08
Sb-125	1.0E-01	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
	1.0E-02	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Te-125m	2.0E-01	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Te-127m	2.0E-01	2.E-07	4.E 06	2.E 07	4.E 07	4.E 08
Te-127	2.0E-01	1.E-08	8.E 07	3.E 08	8.E 08	8.E 09
Te-129m	2.0E-01	3.E-07	4.E 06	1.E 07	4.E 07	4.E 08
Te-129	2.0E-01	4.E-09	3.E 08	9.E 08	3.E 09	3.E 10
Te-131m	2.0E-01	3.E-07	3.E 07	1.E 07	3.E 08	3.E 09
Te-131	2.0E-01	9.E-09	1.E 08	4.E 09	1.E 09	1.E 10
Te-132	2.0E-01	4.E-07	3.E 06	1.E 07	3.E 07	3.E 08
I-125	1.0E-00	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08
I-126	1.0E-00	4.E-07	3.E 06	9.E 06	3.E 07	3.E 08
I-129	1.0E-00	1.E-06	7.E 05	2.E 06	7.E 06	7.E 07
I-131	1.0E-00	3.E-07	4.E 06	1.E 07	4.E 07	4.E 08
I-132	1.0E-00	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
I-133	1.0E-00	5.E-07	2.E 06	7.E 06	2.E 07	2.E 08
I-134	1.0E-00	9.E-09	1.E 08	4.E 08	1.E 09	1.E 10
I-135	1.0E-00	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
Cs-131	1.0E-00	1.E-08	1.E 08	4.E 08	1.E 09	1.E 10
Cs-134m	1.0E-00	2.E-09	6.E 08	2.E 09	6.E 09	6.E 10
Cs-134	1.0E-00	3.E-06	4.E 05	1.E 06	4.E 06	4.E 07
Cs-135	1.0E-00	3.E-07	4.E 06	1.E 07	4.E 07	4.E 08
Cs-136	1.0E-00	4.E-07	2.E 06	8.E 06	2.E 07	2.E 08
Cs-137	1.0E-00	2.E-06	5.E 05	2.E 06	5.E 06	5.E 07
Ba-131	1.0E-01	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Ba-137m	1.0E-01	8.E-11	1.E 10	4.E 10	1.E 11	1.E 12
Ba-140	1.0E-01	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08
La-140	1.0E-03	2.E-07	7.E 06	2.E 07	7.E 07	7.E 08
Ce-141	3.0E-04	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Ce-143	3.0E-04	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Pr-142	3.0E-04	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Pr-143	3.0E-04	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Nd-149	3.0E-04	8.E-09	1.E 08	4.E 08	1.E 09	1.E 10
Pm-149	3.0E-04	8.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Sm-151	3.0E-04	9.E-09	1.E 08	4.E 08	1.E 09	1.E 10
Sm-153	3.0E-04	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Eu-152	1.0E-03	2.E-07	7.E 06	2.E 07	7.E 07	7.E 08
Eu-154	1.0E-03	2.E-07	5.E 06	2.E 07	5.E 07	5.E 08

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Table III-4 - continued

Radionuclide	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
Eu-155	1.0E-03	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Gd-153	3.0E-04	2.E-08	5.E 07	2.E 08	5.E 08	5.E 09
Gd-159	3.0E-04	4.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Tb-160	3.0E-04	1.E-07	8.E 06	3.E 07	8.E 07	8.E 08
Dy-165	3.0E-04	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10
Dy-166	3.0E-04	1.E-07	7.E 06	3.E 07	7.E 07	7.E 08
Ho-166	3.0E-04	1.E-07	1.E 07	3.E 07	1.E 08	1.E 09
Er-169	3.0E-04	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Er-171	3.0E-04	2.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Tm-170	3.0E-04	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
Tm-171	3.0E-04	9.E-09	1.E 08	4.E 08	1.E 09	1.E 10
Yb-175	3.0E-04	4.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Lu-177	3.0E-04	4.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Hf-181	2.0E-03	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Ta-182	1.0E-03	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
W-181	3.0E-01	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10
	1.0E-02	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10
W-185	3.0E-01	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
	1.0E-02	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
W-187	3.0E-01	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
	1.0E-02	5.E-08	2.E 07	7.E 07	2.E 08	2.E 09
Re-183	8.0E-01	4.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Re-186	8.0E-01	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Re-187	8.0E-01	3.E-10	3.E 09	1.E 10	3.E 10	3.E 11
Re-188	8.0E-01	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Os-185	1.0E-02	4.E-08	3.E 07	9.E 07	3.E 08	3.E 09
Os-191m	1.0E-02	7.E-09	1.E 08	5.E 08	1.E 09	1.E 10
Os-191	1.0E-02	5.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Os-193	1.0E-02	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Ir-190	1.0E-02	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Ir-192	1.0E-02	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
Ir-194	1.0E-02	1.E-07	1.E 07	4.E 07	1.E 08	1.E 09
Pt-191	1.0E-02	3.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Pt-193m	1.0E-02	4.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Pt-193	1.0E-02	2.E-09	4.E 08	1.E 09	4.E 09	4.E 10
Pt-197m	1.0E-02	6.E-09	2.E 08	6.E 08	2.E 09	2.E 10

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Table III-4 - continued

Radionuclide[a]	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
Pt-197	1.0E-02	3.E-08	3.E 07	1.E 08	3.E 08	3.E 09
Au-196	1.0E-01	3.E-08	4.E 07	1.E 08	4.E 08	4.E 09
Au-198	1.0E-01	8.E-08	1.E 07	4.E 07	1.E 08	1.E 09
Hg-197	2.0E-02	2.E-08	6.E 07	2.E 08	6.E 08	6.E 09
Hg-203	2.0E-02	5.E-08	2.E 07	8.E 07	2.E 08	2.E 09
Tl-202	1.0E-00	6.E-08	2.E 07	6.E 07	2.E 08	2.E 09
Tl-204	1.0E-00	1.E-07	9.E 06	3.E 07	9.E 07	9.E 08
*Tl-207	1.0E-00	4.E-10	NA[b]	NA	NA	NA
*Tl-208	1.0E-00	6.E-10	NA	NA	NA	NA
Pb-203	2.0E-01	2.E-08	5.E 07	2.E 08	5.E 08	5.E 09
*Pb-210	2.0E-01	7.E-06	NA	NA	NA	NA
*Pb-211	2.0E-01	9.E-09	NA	NA	NA	NA
*Pb-212	2.0E-01	7.E-07	NA	NA	NA	NA
*Pb-214	2.0E-01	7.E-09	NA	NA	NA	NA
Bi-206	5.0E-02	1.E-07	8.E 06	3.E 07	8.E 07	8.E 08
Bi-207	5.0E-02	9.E-08	1.E 07	4.E 07	1.E 08	1.E 09
*Bi-210	5.0E-02	2.E-07	NA	NA	NA	NA
*Bi-211	5.0E-02	4.E-10	NA	NA	NA	NA
*Bi-212	5.0E-02	1.E-08	NA	NA	NA	NA
*Bi-214	5.0E-02	5.E-09	NA	NA	NA	NA
*Po-210	1.0E-01	1.E-05	NA	NA	NA	NA
*Po-212	1.0E-01	8.E-19	NA	NA	NA	NA
*Po-214	1.0E-01	4.E-16	NA	NA	NA	NA
*Po-215	1.0E-01	1.E-14	NA	NA	NA	NA
*Po-216	1.0E-01	2.E-12	NA	NA	NA	NA
*Po-218	1.0E-01	1.E-09	NA	NA	NA	NA
*Ra-223	2.0E-01	1.E-05	NA	NA	NA	NA
*Ra-224	2.0E-01	7.E-06	NA	NA	NA	NA
*Ra-226	2.0E-01	2.E-05	NA	NA	NA	NA
*Ra-228	2.0E-01	1.E-05	NA	NA	NA	NA
*Ac-227	1.0E-03	6.E-05	NA	NA	NA	NA
*Ac-228	1.0E-02	3.E-08	NA	NA	NA	NA
*Th-227	2.0E-04	5.E-07	NA	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

[b] NA = Not available.

Table III-4 - continued

Radionuclide[a]	f_1	Incremental deaths in cohort of 10^5 adults from acute ingestion of 1.0 pCi	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 1 day equal to that from a uniform whole body dose of:			
			50 mrem	170 mrem	500 mrem	5 rem
*Th-228	2.0E-04	3.E-06	NA[b]	NA	NA	NA
*Th-230	2.0E-04	8.E-06	NA	NA	NA	NA
*Th-231	2.0E-04	3.E-08	NA	NA	NA	NA
*Th-232	2.0E-04	7.E-06	NA	NA	NA	NA
*Th-234	2.0E-04	3.E-07	NA	NA	NA	NA
*Pa-231	1.0E-03	7.E-05	NA	NA	NA	NA
Pa-233	1.0E-03	7.E-08	1.E 07	5.E 07	1.E 08	1.E 09
*Pa-234m	1.0E-03	2.E-10	NA	NA	NA	NA
*Pa-234	1.0E-03	4.E-08	NA	NA	NA	NA
*U-234	2.0E-01	1.E-05	NA	NA	NA	NA
	5.0E-02	3.E-06	NA	NA	NA	NA
	2.0E-03	3.E-07	NA	NA	NA	NA
*U-235	2.0E-01	1.E-05	NA	NA	NA	NA
	5.0E-02	3.E-06	NA	NA	NA	NA
	2.0E-03	3.E-07	NA	NA	NA	NA
*U-238	2.0E-01	1.E-05	NA	NA	NA	NA
	5.0E-02	3.E-06	NA	NA	NA	NA
	2.0E-03	3.E-07	NA	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

[b] NA = Not available.

Table III-5 Radionuclide Concentrations in Drinking Water Producing Risk from Ingestion of 2 L/day for Ten Days Equal to That from Reference Radiation Doses

		Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of:			
Radio-nuclide	f_1	50 mrem	170 mrem	500 mrem	5 rem
H-3	1.0E-00	4.E 07	1.E 08	4.E 08	4.E 09
Be-7	5.0E-03	5.E 07	2.E 08	5.E 08	5.E 09
C-14	1.0E-00	1.E 06	4.E 06	1.E 07	1.E 08
F-18	1.0E-00	2.E 07	8.E 07	2.E 08	2.E 09
Na-22	1.0E-00	2.E 05	8.E 05	2.E 06	2.E 07
Na-24	1.0E-00	2.E 06	7.E 06	2.E 07	2.E 08
Si-31	1.0E-02	1.E 07	4.E 07	1.E 08	1.E 09
P-32	8.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
S-35	8.0E-01	7.E 06	2.E 07	7.E 07	7.E 08
	1.0E-01	7.E 06	2.E 07	7.E 07	7.E 08
Cl-36	1.0E-00	8.E 05	3.E 06	8.E 06	8.E 07
Cl-38	1.0E-00	1.E 07	5.E 07	1.E 08	1.E 09
K-40	1.0E-00	1.E 05	5.E 05	1.E 06	1.E 07
K-42	1.0E-00	2.E 06	8.E 06	2.E 07	2.E 08
Ca-45	3.0E-01	1.E 06	3.E 06	1.E 07	1.E 08
Ca-47	3.0E-01	7.E 05	2.E 06	7.E 06	7.E 07
Sc-46	1.0E-04	1.E 06	3.E 06	1.E 07	1.E 08
Sc-47	1.0E-04	2.E 06	8.E 06	2.E 07	2.E 08
Sc-48	1.0E-04	9.E 05	3.E 06	9.E 06	9.E 07
V-48	1.0E-02	7.E 05	2.E 06	7.E 06	7.E 07
Cr-51	1.0E-01	4.E 07	1.E 08	4.E 08	4.E 09
	1.0E-02	4.E 07	1.E 08	4.E 08	4.E 09
Mn-52	1.0E-01	7.E 05	3.E 06	7.E 06	7.E 07
Mn-54	1.0E-01	1.E 06	5.E 06	1.E 07	1.E 08
Mn-56	1.0E-01	6.E 06	2.E 07	6.E 07	6.E 08
Fe-55	1.0E-01	5.E 06	2.E 07	5.E 07	5.E 08
Fe-59	1.0E-01	6.E 05	2.E 06	6.E 06	6.E 07
Co-57	3.0E-01	3.E 06	9.E 06	3.E 07	3.E 08
	5.0E-02	6.E 06	2.E 07	6.E 07	6.E 08
Co-58m	3.0E-01	6.E 07	2.E 08	6.E 08	6.E 09
	5.0E-02	6.E 07	2.E 08	6.E 08	6.E 09
Co-58	3.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
	5.0E-02	2.E 06	6.E 06	2.E 07	2.E 08
Co-60	3.0E-01	1.E 05	4.E 06	1.E 06	1.E 07
	5.0E-02	4.E 05	1.E 05	4.E 06	4.E 07
Ni-59	5.0E-02	2.E 07	5.E 07	2.E 08	2.E 09

. - continued

Table III-5 - continued

Radionuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of:			
		50 mrem	170 mrem	500 mrem	5 rem
Ni-63	5.0E-02	6.E 06	2.E 07	6.E 07	6.E 08
Ni-65	5.0E-02	1.E 07	3.E 07	1.E 08	1.E 09
Cu-64	5.0E-01	1.E 07	4.E 07	1.E 08	1.E 09
Zn-65	5.0E-01	2.E 05	6.E 05	2.E 06	2.E 07
Zn-69m	5.0E-01	4.E 06	1.E 07	4.E 07	4.E 08
Zn-69	5.0E-01	6.E 07	2.E 08	6.E 08	6.E 09
Ga-72	1.0E-03	1.E 06	5.E 06	1.E 07	1.E 08
Ge-71	1.0E-00	3.E 08	1.E 09	3.E 09	3.E 10
As-73	5.0E-01	6.E 06	2.E 07	6.E 07	6.E 08
As-74	5.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
As-76	5.0E-01	9.E 05	3.E 06	9.E 06	9.E 07
As-77	5.0E-01	4.E 06	1.E 07	4.E 07	4.E 08
Se-75	8.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
	5.0E-02	3.E 06	9.E 06	3.E 07	3.E 08
Br-82	1.0E-00	2.E 06	5.E 06	2.E 07	2.E 08
Rb-86	1.0E-00	3.E 05	9.E 06	3.E 06	3.E 07
Rb-87	1.0E-00	3.E 05	1.E 06	3.E 06	3.E 07
Sr-85m	3.0E-01	2.E 08	7.E 08	2.E 09	2.E 10
	1.0E-02	2.E 08	7.E 08	2.E 09	2.E 10
Sr-85	3.0E-01	2.E 06	7.E 06	2.E 07	2.E 08
	1.0E-02	4.E 06	2.E 07	4.E 07	4.E 08
Sr-89	3.0E-01	4.E 05	1.E 06	4.E 06	4.E 07
	1.0E-02	5.E 05	2.E 06	5.E 06	5.E 07
Sr-90	3.0E-01	2.E 04	8.E 04	2.E 05	2.E 06
	1.0E-02	3.E 05	1.E 06	3.E 06	3.E 07
Sr-91	3.0E-01	2.E 06	7.E 06	2.E 07	2.E 08
	1.0E-02	2.E 06	7.E 06	2.E 07	2.E 08
Sr-92	3.0E-01	3.E 06	1.E 07	3.E 07	3.E 08
	1.0E-02	3.E 06	1.E 07	3.E 07	3.E 08
Y-90	1.0E-04	5.E 05	2.E 06	5.E 06	5.E 07
Y-91m	1.0E-04	1.E 08	4.E 08	1.E 09	1.E 10
Y-91	1.0E-04	5.E 05	2.E 06	5.E 06	5.E 07
Y-92	1.0E-04	3.E 06	1.E 07	3.E 07	3.E 08
Y-93	1.0E-04	1.E 06	4.E 06	1.E 07	1.E 08
Zr-93	2.0E-03	5.E 06	2.E 07	5.E 07	5.E 08
Zr-95	2.0E-03	2.E 06	5.E 06	2.E 07	2.E 08
Zr-97	2.0E-03	7.E 05	2.E 06	7.E 06	7.E 07
Nb-93m	1.0E-02	1.E 07	3.E 07	1.E 08	1.E 09
Nb-95m	1.0E-02	2.E 06	8.E 06	2.E 07	2.E 08
Nb-95	1.0E-02	2.E 06	8.E 06	2.E 07	2.E 08
Nb-97m	1.0E-02	1.E 09	4.E 09	1.E 10	1.E 11

- continued

Table III-5 - continued

Radio-nuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
Nb-97	1.0E-02	2.E 07	8.E 07	2.E 08	2.E 09
Mo-99	8.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Mo-99	5.0E-02	1.E 06	3.E 06	1.E 07	1.E 08
Tc-96m	8.0E-01	1.E 08	4.E 08	1.E 09	1.E 10
Tc-96	8.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Tc-97m	8.0E-01	3.E 06	9.E 06	3.E 07	3.E 08
Tc-97	8.0E-01	2.E 07	6.E 07	2.E 08	2.E 09
Tc-99m	8.0E-01	6.E 07	2.E 08	6.E 08	6.E 09
Tc-99	8.0E-01	2.E 06	7.E 06	2.E 07	2.E 08
Ru-97	5.0E-02	9.E 06	3.E 07	9.E 07	9.E 08
Ru-103	5.0E-02	2.E 06	6.E 06	2.E 07	2.E 08
Ru-106	5.0E-02	2.E 05	5.E 05	2.E 06	2.E 07
Rh-103m	5.0E-02	5.E 08	2.E 09	5.E 09	5.E 10
Rh-105	5.0E-02	4.E 06	1.E 07	4.E 07	4.E 08
Rh-106	5.0E-02	8.E 08	3.E 09	8.E 09	8.E 10
Pd-103	5.0E-03	7.E 06	2.E 07	7.E 07	7.E 08
Pd-109	5.0E-03	2.E 06	8.E 06	2.E 07	2.E 08
Ag-105	5.0E-02	2.E 06	7.E 06	2.E 07	2.E 08
Ag-109m	5.0E-02	1.E 10	4.E 10	1.E 11	1.E 12
Ag-110m	5.0E-02	3.E 05	1.E 06	3.E 06	3.E 07
Ag-110	5.0E-02	1.E 09	4.E 09	1.E 10	1.E 11
Ag-111	5.0E-02	9.E 05	3.E 06	9.E 06	9.E 07
Cd-109	5.0E-02	2.E 05	7.E 05	2.E 06	2.E 07
Cd-115m	5.0E-02	3.E 05	1.E 06	3.E 06	3.E 07
Cd-115	5.0E-02	9.E 05	3.E 06	9.E 06	9.E 07
In-113m	2.0E-02	6.E 07	2.E 08	6.E 08	6.E 09
In-114m	2.0E-02	3.E 05	9.E 05	3.E 06	3.E 07
In-114	2.0E-02	6.E 08	2.E 09	6.E 09	6.E 10
In-115m	2.0E-02	2.E 07	6.E 07	2.E 08	2.E 09
In-115	2.0E-02	3.E 04	1.E 05	3.E 05	3.E 06
Sn-113	2.0E-02	2.E 06	6.E 06	2.E 07	2.E 08
Sn-125	2.0E-02	4.E 05	1.E 06	4.E 06	4.E 07
Sb-122	1.0E-01	8.E 05	3.E 06	8.E 06	8.E 07
	1.0E-02	7.E 05	2.E 06	7.E 06	7.E 07
Sb-124	1.0E-01	5.E 05	2.E 06	5.E 06	5.E 07
	1.0E-02	6.E 05	2.E 06	6.E 06	6.E 07
Sb-125	1.0E-01	2.E 06	6.E 06	2.E 07	2.E 08
	1.0E-02	2.E 06	7.E 06	2.E 07	2.E 08
Te-125m	2.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Te-127m	2.0E-01	4.E 05	2.E 06	4.E 06	4.E 07
Te-127	2.0E-01	8.E 06	3.E 07	8.E 07	8.E 08

- continued

Table III-5 - continued

Radio-nuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
Te-129m	2.0E-01	4.E 05	1.E 06	4.E 06	4.E 07
Te-129	2.0E-01	3.E 07	9.E 07	3.E 08	3.E 09
Te-131m	2.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
Te-131	2.0E-01	1.E 07	4.E 07	1.E 08	1.E 09
Te-132	2.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
I-125	1.0E-00	5.E 05	2.E 06	5.E 06	5.E 07
I-126	1.0E-00	3.E 05	9.E 05	3.E 06	3.E 07
I-129	1.0E-00	7.E 04	2.E 05	7.E 05	7.E 06
I-131	1.0E-00	4.E 05	1.E 06	4.E 06	4.E 07
I-132	1.0E-00	3.E 06	1.E 07	4.E 07	3.E 08
I-133	1.0E-00	2.E 05	7.E 05	2.E 06	2.E 07
I-134	1.0E-00	1.E 07	4.E 07	1.E 08	1.E 09
I-135	1.0E-00	9.E 05	3.E 06	9.E 06	9.E 07
Cs-131	1.0E-00	1.E 07	4.E 07	1.E 08	1.E 09
Cs-134m	1.0E-00	6.E 07	2.E 08	6.E 08	6.E 09
Cs-134	1.0E-00	4.E 04	1.E 05	4.E 05	4.E 06
Cs-135	1.0E-00	4.E 05	1.E 06	4.E 06	4.E 07
Cs-136	1.0E-00	2.E 05	8.E 05	2.E 06	2.E 07
Cs-137	1.0E-00	5.E 04	2.E 05	5.E 05	5.E 06
Ba-131	1.0E-01	3.E 06	1.E 07	3.E 07	3.E 08
Ba-137m	1.0E-01	1.E 09	4.E 09	1.E 10	1.E 11
Ba-140	1.0E-01	5.E 05	2.E 06	5.E 06	5.E 07
La-140	1.0E-03	7.E 05	2.E 06	7.E 06	7.E 07
Ce-141	3.0E-04	2.E 06	6.E 06	2.E 07	2.E 08
Ce-143	3.0E-04	1.E 06	4.E 06	1.E 07	1.E 08
Pr-142	3.0E-04	1.E 06	4.E 06	1.E 07	1.E 08
Pr-143	3.0E-04	1.E 06	4.E 06	1.E 07	1.E 08
Nd-149	3.0E-04	1.E 07	4.E 07	1.E 08	1.E 09
Pm-149	3.0E-04	1.E 06	4.E 06	1.E 07	1.E 08
Sm-151	3.0E-04	1.E 07	4.E 07	1.E 08	1.E 09
Sm-153	3.0E-04	2.E 06	6.E 06	2.E 07	2.E 08
Eu-152	1.0E-03	7.E 05	2.E 06	7.E 06	7.E 07
Eu-154	1.0E-03	5.E 05	2.E 06	5.E 06	5.E 07
Eu-155	1.0E-03	3.E 06	1.E 07	3.E 07	3.E 08
Gd-153	3.0E-04	5.E 06	2.E 07	5.E 07	5.E 08
Gd-159	3.0E-04	3.E 06	1.E 07	3.E 07	3.E 08
Tb-160	3.0E-04	8.E 05	3.E 06	8.E 06	8.E 07
Dy-165	3.0E-04	2.E 07	6.E 07	2.E 08	2.E 09
Dy-166	3.0E-04	7.E 05	3.E 06	7.E 06	7.E 07

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Table III-5 - continued

Radio-nuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
Ho-166	3.0E-04	1.E 06	3.E 06	1.E 07	1.E 08
Er-169	3.0E-04	3.E 06	1.E 07	3.E 07	3.E 08
Er-171	3.0E-04	4.E 06	1.E 07	4.E 07	4.E 08
Tm-170	3.0E-04	9.E 05	3.E 06	9.E 06	9.E 07
Tm-171	3.0E-04	1.E 07	4.E 07	1.E 08	1.E 09
Yb-175	3.0E-04	3.E 06	1.E 07	3.E 07	3.E 08
Lu-177	3.0E-04	2.E 06	8.E 06	2.E 07	2.E 08
Hf-181	2.0E-03	1.E 06	4.E 06	1.E 07	1.E 08
Ta-182	1.0E-03	9.E 05	3.E 06	9.E 06	9.E 07
W-181	3.0E-01	2.E 07	6.E 07	2.E 08	2.E 09
	1.0E-02	2.E 07	6.E 07	2.E 08	2.E 09
W-185	3.0E-01	3.E 06	1.E 07	3.E 07	3.E 08
	1.0E-02	3.E 06	9.E 06	3.E 07	3.E 08
W-187	3.0E-01	3.E 06	9.E 06	3.E 07	3.E 08
	1.0E-02	2.E 06	7.E 06	2.E 07	2.E 08
Re-183	8.0E-01	3.E 06	1.E 07	3.E 07	3.E 08
Re-186	8.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Re-187	8.0E-01	3.E 08	1.E 09	3.E 09	3.E 10
Re-188	8.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Os-185	1.0E-02	3.E 06	9.E 06	3.E 07	3.E 08
Os-191m	1.0E-02	1.E 07	5.E 07	1.E 08	1.E 09
Os-191	1.0E-02	2.E 06	8.E 06	2.E 07	2.E 08
Os-193	1.0E-02	2.E 06	6.E 06	2.E 07	2.E 08
Ir-190	1.0E-02	1.E 06	4.E 06	1.E 07	1.E 08
Ir-192	1.0E-02	9.E 05	3.E 06	9.E 06	9.E 07
Ir-194	1.0E-02	1.E 06	4.E 06	1.E 07	1.E 08
Pt-191	1.0E-02	4.E 06	1.E 07	4.E 07	4.E 08
Pt-193m	1.0E-02	3.E 06	1.E 07	3.E 07	3.E 08
Pt-193	1.0E-02	4.E 07	1.E 08	4.E 08	4.E 09
Pt-197m	1.0E-02	2.E 07	6.E 07	2.E 08	2.E 09
Pt-197	1.0E-02	3.E 06	1.E 07	3.E 07	3.E 08
Au-196	1.0E-01	4.E 06	1.E 07	4.E 07	4.E 08
Au-198	1.0E-01	1.E 06	4.E 06	1.E 07	1.E 08
Hg-197	2.0E-02	6.E 06	2.E 07	6.E 07	6.E 08
Hg-203	2.0E-02	2.E 06	8.E 06	2.E 07	2.E 08
Tl-202	1.0E-00	2.E 06	6.E 06	2.E 07	2.E 08
Tl-204	1.0E-00	9.E 05	3.E 06	9.E 06	9.E 07

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Table III-5 - continued

		Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of:			
Radio-nuclide[a]	f_1	50 mrem	170 mrem	500 mrem	5 rem
*Tl-207	1.0E-00	NA[b]	NA	NA	NA
*Tl-208	9.5E-01	NA	NA	NA	NA
Pb-203	2.0E-01	5.E 06	2.E 07	5.E 07	5.E 08
*Pb-210	2.0E-01	NA	NA	NA	NA
*Pb-211	2.0E-01	NA	NA	NA	NA
*Pb-212	2.0E-01	NA	NA	NA	NA
*Pb-214	2.0E-01	NA	NA	NA	NA
Bi-206	5.0E-02	8.E 05	3.E 06	8.E 06	8.E 07
Bi-207	5.0E-02	1.E 06	4.E 06	1.E 07	1.E 08
*Bi-210	5.0E-02	NA	NA	NA	NA
*Bi-211	5.0E-02	NA	NA	NA	NA
*Bi-212	5.0E-02	NA	NA	NA	NA
*Bi-214	5.0E-02	NA	NA	NA	NA
*Po-210	1.0E-01	NA	NA	NA	NA
*Po-212	1.0E-01	NA	NA	NA	NA
*Po-214	1.0E-01	NA	NA	NA	NA
*Po-215	1.0E-01	NA	NA	NA	NA
*Po-216	1.0E-01	NA	NA	NA	NA
*Po-218	1.0E-01	NA	NA	NA	NA
*Ra-223	2.0E-01	NA	NA	NA	NA
*Ra-224	2.0E-01	NA	NA	NA	NA
*Ra-226	2.0E-01	NA	NA	NA	NA
*Ra-228	2.0E-01	NA	NA	NA	NA
*Ac-227	1.0E-03	NA	NA	NA	NA
*Ac-228	1.0E-02	NA	NA	NA	NA
*Th-227	2.0E-04	NA	NA	NA	NA
*Th-228	2.0E-04	NA	NA	NA	NA
*Th-230	2.0E-04	NA	NA	NA	NA
*Th-231	2.0E-04	NA	NA	NA	NA
*Th-232	2.0E-04	NA	NA	NA	NA
*Th-234	2.0E-04	NA	NA	NA	NA
*Pa-231	1.0E-03	NA	NA	NA	NA
Pa-233	1.0E-03	1.E 06	5.E 06	1.E 07	1.E 08
*Pa-234m	1.0E-03	NA	NA	NA	NA
*Pa-234	1.0E-03	NA	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

[b] NA = Not available.

Table III-5 - continued

Radio-nuclide[a]	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 10 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
*U-234	2.0E-01	NA[b]	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA
*U-235	2.0E-01	NA	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA
*U-238	2.0E-01	NA	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated with an *.

[b] NA = Not available.

Table III-6 Radionuclide Concentrations in Drinking Water Producing Risk from Ingestion of 2 L/day for 365 Days Equal to That from Reference Radiation Doses

Radionuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
H-3	1.0E-00	1.E 06	4.E 06	1.E 07	1.E 08
Be-7	5.0E-03	2.E 06	5.E 06	2.E 07	2.E 08
C-14	1.0E-00	3.E 04	1.E 05	3.E 05	3.E 06
F-18	1.0E-00	7.E 05	2.E 06	7.E 06	7.E 07
Na-22	1.0E-00	6.E 03	2.E 04	6.E 04	6.E 05
Na-24	1.0E-00	5.E 04	2.E 05	5.E 05	5.E 06
Si-31	1.0E-02	3.E 05	1.E 06	3.E 06	3.E 07
P-32	8.0E-01	9.E 03	3.E 04	9.E 04	9.E 05
S-35	1.0E-01	2.E 05	6.E 05	2.E 06	2.E 07
S-35	8.0E-01	2.E 05	6.E 05	2.E 06	2.E 07
Cl-36	9.5E-01	2.E 04	8.E 04	2.E 05	2.E 06
Cl-38	9.5E-01	4.E 05	1.E 06	4.E 06	4.E 07
K-40	9.5E-01	4.E 03	1.E 04	4.E 04	4.E 05
K-42	9.5E-01	7.E 04	2.E 05	7.E 05	7.E 06
Ca-45	3.0E-01	3.E 04	9.E 04	3.E 05	3.E 06
Ca-47	3.0E-01	2.E 04	6.E 04	2.E 05	2.E 06
Sc-46	1.0E-04	3.E 04	9.E 04	3.E 05	3.E 06
Sc-47	1.0E-04	7.E 04	2.E 05	7.E 05	7.E 06
Sc-48	1.0E-04	2.E 04	8.E 04	2.E 05	2.E 06
V-48	1.0E-02	2.E 04	7.E 04	2.E 05	2.E 06
Cr-51	1.0E-01	1.E 06	3.E 06	1.E 07	1.E 08
Cr-51	1.0E-02	1.E 06	4.E 06	1.E 07	1.E 08
Mn-52	1.0E-01	2.E 04	7.E 04	2.E 05	2.E 06
Mn-54	1.0E-01	4.E 04	1.E 05	4.E 05	4.E 06
Mn-56	1.0E-01	2.E 05	6.E 05	2.E 06	2.E 07
Fe-55	1.0E-01	1.E 05	5.E 05	1.E 06	1.E 07
Fe-59	1.0E-01	2.E 04	5.E 04	2.E 05	2.E 06
Co-57	3.0E-01	8.E 04	3.E 05	8.E 05	8.E 06
	5.0E-02	2.E 05	6.E 05	2.E 06	2.E 07
Co-58m	3.0E-01	2.E 06	5.E 06	2.E 07	2.E 08
	5.0E-02	2.E 06	6.E 06	2.E 07	2.E 08
Co-58	3.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
	5.0E-02	5.E 04	2.E 05	5.E 05	5.E 06
Co-60	3.0E-01	3.E 03	1.E 04	3.E 04	3.E 05
	5.0E-02	1.E 04	4.E 04	1.E 05	1.E 06
Ni-59	5.0E-02	4.E 05	2.E 06	4.E 06	4.E 07

- continued

Table III-6 - continued

		Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of:			
Radio-nuclide	f_1	50 mrem	170 mrem	500 mrem	5 rem
Ni-63	5.0E-02	2.E 05	6.E 05	2.E 06	2.E 07
Ni-65	5.0E-02	3.E 05	9.E 05	3.E 06	3.E 07
Cu-64	5.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
Zn-65	5.0E-01	5.E 03	2.E 04	5.E 04	5.E 05
Zn-69m	5.0E-01	1.E 05	4.E 05	1.E 06	1.E 07
Zn-69	5.0E-01	2.E 06	5.E 06	2.E 07	2.E 08
Ga-72	1.0E-03	4.E 04	1.E 05	4.E 05	4.E 06
Ge-71	1.0E-00	8.E 06	3.E 07	8.E 07	8.E 08
As-73	5.0E-01	2.E 05	5.E 05	2.E 06	2.E 07
As-74	5.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
As-76	5.0E-01	3.E 04	9.E 04	3.E 05	3.E 06
As-77	5.0E-01	1.E 05	3.E 05	1.E 06	1.E 07
Se-75	8.0E-01	8.E 03	3.E 04	8.E 04	8.E 05
	5.0E-02	7.E 04	2.E 05	7.E 05	7.E 06
Br-82	1.0E-00	4.E 04	1.E 05	4.E 05	4.E 06
Rb-86	1.0E-00	8.E 03	3.E 04	8.E 04	8.E 05
Rb-87	1.0E-00	8.E 03	3.E 04	8.E 04	8.E 05
Sr-85m	3.0E-01	6.E 06	2.E 07	6.E 07	6.E 08
	1.0E-02	6.E 06	2.E 07	6.E 07	6.E 08
Sr-85	3.0E-01	5.E 04	2.E 05	5.E 05	5.E 06
	1.0E-02	1.E 05	4.E 05	1.E 06	1.E 07
Sr-89	3.0E-01	1.E 04	4.E 04	1.E 05	1.E 06
	1.0E-02	1.E 04	5.E 04	1.E 05	1.E 06
Sr-90	3.0E-01	7.E 02	2.E 03	7.E 03	7.E 04
	1.0E-02	9.E 03	3.E 04	9.E 04	9.E 05
Sr-91	3.0E-01	6.E 04	2.E 05	6.E 05	6.E 06
	1.0E-02	5.E 04	2.E 05	5.E 05	5.E 06
Sr-92	3.0E-01	9.E 04	3.E 05	9.E 05	9.E 06
	1.0E-02	9.E 04	3.E 05	9.E 05	9.E 06
Y-90	1.0E-04	1.E 04	5.E 04	1.E 05	1.E 06
Y-91m	1.0E-04	3.E 06	1.E 07	3.E 07	3.E 08
Y-91	1.0E-04	1.E 04	5.E 04	1.E 05	1.E 06
Y-92	1.0E-04	9.E 04	3.E 05	9.E 05	9.E 06
Y-93	1.0E-04	4.E 04	1.E 05	4.E 05	4.E 06
Zr-93	2.0E-03	1.E 05	4.E 05	1.E 06	1.E 07
Zr-95	2.0E-03	4.E 04	1.E 05	4.E 05	4.E 06
Zr-97	2.0E-03	2.E 04	6.E 04	2.E 05	2.E 06
Nb-93m	1.0E-02	3.E 05	9.E 05	3.E 06	3.E 07
Nb-95m	1.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Nb-95	1.0E-02	7.E 04	2.E 05	7.E 05	7.E 06
Nb-97m	1.0E-02	3.E 07	1.E 08	3.E 08	3.E 09

- continued

Table III-6 - continued

Radio-nuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
Nb-97	1.0E-02	6.E 05	2.E 06	6.E 06	6.E 07
Mo-99	8.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
Mo-99	5.0E-02	3.E 04	1.E 05	3.E 05	3.E 06
Tc-96m	8.0E-01	3.E 06	1.E 07	3.E 07	3.E 08
Tc-96	8.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
Tc-97m	8.0E-01	7.E 04	2.E 05	7.E 05	7.E 06
Tc-97	8.0E-01	5.E 05	2.E 06	5.E 06	5.E 07
Tc-99m	8.0E-01	2.E 06	5.E 06	2.E 07	2.E 08
Tc-99	8.0E-01	6.E 04	2.E 05	6.E 05	6.E 06
Ru-97	5.0E-02	3.E 05	9.E 05	3.E 06	3.E 07
Ru-103	5.0E-02	5.E 04	2.E 05	5.E 05	5.E 06
Ru-106	5.0E-02	4.E 03	1.E 04	4.E 04	4.E 05
Rh-103m	5.0E-02	1.E 07	4.E 07	1.E 08	1.E 09
Rh-105	5.0E-02	1.E 05	3.E 05	1.E 06	1.E 07
Rh-106	5.0E-02	2.E 07	7.E 07	2.E 08	2.E 09
Pd-103	5.0E-03	2.E 05	6.E 05	2.E 06	2.E 07
Pd-109	5.0E-03	6.E 04	2.E 05	6.E 05	6.E 06
Ag-105	5.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Ag-109m	5.0E-02	3.E 08	1.E 09	3.E 09	3.E 10
Ag-110m	5.0E-02	9.E 03	3.E 04	9.E 04	9.E 05
Ag-110	5.0E-02	3.E 07	1.E 08	3.E 08	3.E 09
Ag-111	5.0E-02	3.E 04	9.E 04	3.E 05	3.E 06
Cd-109	5.0E-02	6.E 03	2.E 04	6.E 04	6.E 05
Cd-115m	5.0E-02	9.E 03	3.E 04	9.E 04	9.E 05
Cd-115	5.0E-02	3.E 04	9.E 04	3.E 05	3.E 06
In-113m	2.0E-02	2.E 06	5.E 06	2.E 07	2.E 08
In-114m	2.0E-02	7.E 03	2.E 04	7.E 04	7.E 05
In-114	2.0E-02	2.E 07	6.E 07	2.E 08	2.E 09
In-115m	2.0E-02	5.E 05	2.E 06	5.E 06	5.E 07
In-115	2.0E-02	8.E 02	3.E 03	8.E 03	8.E 04
Sn-113	2.0E-02	5.E 04	2.E 05	5.E 05	5.E 06
Sn-125	2.0E-02	1.E 04	4.E 04	1.E 05	1.E 06
Sb-122	1.0E-01	2.E 04	7.E 04	2.E 05	2.E 06
Sb-122	1.0E-02	2.E 04	7.E 04	2.E 05	2.E 06
Sb-124	1.0E-01	1.E 04	5.E 04	1.E 05	1.E 06
Sb-124	1.0E-02	2.E 04	5.E 04	2.E 05	2.E 06
Sb-125	1.0E-01	5.E 04	2.E 05	5.E 05	5.E 06
Sb-125	1.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Te-125m	2.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
Te-127m	2.0E-01	1.E 04	4.E 04	1.E 05	1.E 06
Te-127	2.0E-01	2.E 05	8.E 05	2.E 06	2.E 07

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Table III-6 - continued

Radio-nuclide	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
Te-129m	2.0E-01	1.E 04	4.E 04	1.E 05	1.E 06
Te-129	2.0E-01	7.E 05	3.E 06	7.E 06	7.E 07
Te-131m	2.0E-01	9.E 03	3.E 04	9.E 04	9.E 05
Te-131	2.0E-01	3.E 05	1.E 06	3.E 06	3.E 07
Te-132	2.0E-01	8.E 03	3.E 04	8.E 04	8.E 05
I-125	1.0E-00	1.E 04	5.E 04	1.E 05	1.E 06
I-126	1.0E-00	7.E 03	2.E 04	7.E 04	7.E 05
I-129	1.0E-00	2.E 03	7.E 03	2.E 04	2.E 05
I-131	1.0E-00	1.E 04	3.E 04	1.E 05	1.E 06
I-132	1.0E-00	9.E 04	3.E 05	9.E 05	9.E 06
I-133	1.0E-00	5.E 03	2.E 04	5.E 04	5.E 05
I-134	1.0E-00	3.E 05	1.E 06	3.E 06	3.E 07
I-135	1.0E-00	3.E 04	9.E 04	3.E 05	3.E 06
Cs-131	1.0E-00	3.E 05	1.E 06	3.E 06	3.E 07
Cs-134m	1.0E-00	2.E 06	6.E 06	2.E 07	2.E 08
Cs-134	1.0E-00	1.E 03	3.E 03	1.E 04	1.E 05
Cs-135	1.0E-00	1.E 04	3.E 04	1.E 05	1.E 06
Cs-136	1.0E-00	6.E 03	2.E 04	6.E 04	6.E 05
Cs-137	1.0E-00	1.E 03	5.E 03	1.E 04	1.E 05
Ba-131	1.0E-01	9.E 04	3.E 05	9.E 05	9.E 06
Ba-137m	1.0E-01	4.E 07	1.E 08	4.E 08	4.E 09
Ba-140	1.0E-01	1.E 04	5.E 04	1.E 05	1.E 06
La-140	1.0E-03	2.E 04	6.E 04	2.E 05	2.E 06
Ce-141	3.0E-04	5.E 04	2.E 05	5.E 05	5.E 06
Ce-143	3.0E-04	3.E 04	1.E 05	3.E 05	3.E 06
Pr-142	3.0E-04	3.E 04	1.E 05	3.E 05	3.E 06
Pr-143	3.0E-04	3.E 04	1.E 05	3.E 05	3.E 06
Nd-149	3.0E-04	3.E 05	1.E 06	3.E 06	3.E 07
Pm-149	3.0E-04	4.E 04	1.E 05	4.E 05	4.E 06
Sm-151	3.0E-04	3.E 05	1.E 06	3.E 06	3.E 07
Sm-153	3.0E-04	5.E 04	2.E 05	5.E 05	5.E 06
Eu-152	1.0E-03	2.E 04	6.E 04	2.E 05	2.E 06
Eu-154	1.0E-03	1.E 04	4.E 04	1.E 05	1.E 06
Eu-155	1.0E-03	8.E 04	3.E 05	8.E 05	8.E 06
Gd-153	3.0E-04	1.E 05	4.E 05	1.E 06	1.E 07
Gd-159	3.0E-04	8.E 04	3.E 05	8.E 05	8.E 06
Tb-160	3.0E-04	2.E 04	8.E 04	2.E 05	2.E 06
Dy-165	3.0E-04	5.E 05	2.E 06	5.E 06	5.E 07
Dy-166	3.0E-04	2.E 04	7.E 04	2.E 05	2.E 06
Ho-166	3.0E-04	3.E 04	9.E 04	3.E 05	3.E 06
Er-169	3.0E-04	9.E 04	3.E 05	9.E 05	9.E 06

- continued

Table III-6 - continued

Radio-nuclide[a]	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of:			
		50 mrem	170 mrem	500 mrem	5 rem
Er-171	3.0E-04	1.E 05	4.E 05	1.E 06	1.E 07
Tm-170	3.0E-04	3.E 04	9.E 04	3.E 05	3.E 06
Tm-171	3.0E-04	3.E 05	1.E 06	3.E 06	3.E 07
Yb-175	3.0E-04	8.E 04	3.E 05	8.E 05	8.E 06
Lu-177	3.0E-04	7.E 04	2.E 05	7.E 05	7.E 06
Hf-181	2.0E-03	3.E 04	1.E 05	3.E 05	3.E 06
Ta-182	1.0E-03	2.E 04	8.E 04	2.E 05	2.E 06
W-181	3.0E-01	5.E 05	2.E 06	5.E 06	5.E 07
	1.0E-02	5.E 05	2.E 06	5.E 06	5.E 07
W-185	3.0E-01	9.E 04	3.E 05	9.E 05	9.E 06
	1.0E-02	7.E 04	2.E 05	7.E 05	7.E 06
W-187	3.0E-01	7.E 04	2.E 05	7.E 05	7.E 06
	1.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Re-183	8.0E-01	8.E 04	3.E 05	8.E 05	8.E 06
Re-186	8.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
Re-187	8.0E-01	9.E 06	3.E 07	9.E 07	9.E 08
Re-188	8.0E-01	3.E 04	9.E 04	3.E 05	3.E 06
Os-185	1.0E-02	7.E 04	2.E 05	7.E 05	7.E 06
Os-191m	1.0E-02	4.E 05	1.E 06	4.E 06	4.E 07
Os-191	1.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Os-193	1.0E-02	5.E 04	2.E 05	5.E 05	5.E 06
Ir-190	1.0E-02	3.E 04	1.E 05	3.E 05	3.E 06
Ir-192	1.0E-02	3.E 04	9.E 04	3.E 05	3.E 06
Ir-194	1.0E-02	3.E 04	1.E 05	3.E 05	3.E 06
Pt-191	1.0E-02	1.E 05	4.E 05	1.E 06	1.E 07
Pt-193m	1.0E-02	8.E 04	3.E 05	8.E 05	8.E 06
Pt-193	1.0E-02	1.E 06	4.E 06	1.E 07	1.E 08
Pt-197m	1.0E-02	5.E 05	2.E 06	5.E 06	5.E 07
Pt-197	1.0E-02	9.E 04	3.E 05	9.E 05	9.E 06
Au-196	1.0E-01	1.E 05	4.E 05	1.E 06	1.E 07
Au-198	1.0E-01	3.E 04	1.E 05	3.E 05	3.E 06
Hg-197	2.0E-02	2.E 05	5.E 05	2.E 06	2.E 07
Hg-203	2.0E-02	6.E 04	2.E 05	6.E 05	6.E 06
Tl-202	1.0E-00	5.E 04	2.E 05	5.E 05	5.E 06
Tl-204	1.0E-00	3.E 04	9.E 04	3.E 05	3.E 06
*Tl-207	1.0E-00	NA[b]	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated by an *.

[b] NA = Not available.

Table III-6 - continued

Radio-nuclide[a]	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
*Tl-208	1.0E-00	NA[b]	NA	NA	NA
Pb-203	2.0E-01	1.E 05	5.E 05	1.E 06	1.E 07
*Pb-210	2.0E-01	NA	NA	NA	NA
*Pb-211	2.0E-01	NA	NA	NA	NA
*Pb-212	2.0E-01	NA	NA	NA	NA
*Pb-214	2.0E-01	NA	NA	NA	NA
Bi-206	5.0E-02	2.E 04	7.E 04	2.E 05	2.E 06
Bi-207	5.0E-02	3.E 04	1.E 05	3.E 05	3.E 06
*Bi-210	5.0E-02	NA	NA	NA	NA
*Bi-211	5.0E-02	NA	NA	NA	NA
*Bi-212	5.0E-02	NA	NA	NA	NA
*Bi-214	5.0E-02	NA	NA	NA	NA
*Po-210	1.0E-01	NA	NA	NA	NA
*Po-212	1.0E-01	NA	NA	NA	NA
*Po-214	1.0E-01	NA	NA	NA	NA
*Po-215	1.0E-01	NA	NA	NA	NA
*Po-216	1.0E-01	NA	NA	NA	NA
*Po-218	1.0E-01	NA	NA	NA	NA
*Ra-223	2.0E-01	NA	NA	NA	NA
*Ra-224	2.0E-01	NA	NA	NA	NA
*Ra-226	2.0E-01	NA	NA	NA	NA
*Ra-228	2.0E-01	NA	NA	NA	NA
*Ac-227	1.0E-03	NA	NA	NA	NA
*Ac-228	1.0E-02	NA	NA	NA	NA
*Th-227	2.0E-04	NA	NA	NA	NA
*Th-228	2.0E-04	NA	NA	NA	NA
*Th-230	2.0E-04	NA	NA	NA	NA
*Th-231	2.0E-04	NA	NA	NA	NA
*Th-232	2.0E-04	NA	NA	NA	NA
*Th-234	2.0E-04	NA	NA	NA	NA
*Pa-231	1.0E-03	NA	NA	NA	NA
Pa-233	1.0E-03	4.E 04	1.E 05	4.E 05	4.E 06
*Pa-234m	1.0E-03	NA	NA	NA	NA
*Pa-234	1.0E-03	NA	NA	NA	NA

- continued

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[a] Radionuclides that are naturally-occurring are designated by an *.
[b] NA = Not available.

Table III-6 - continued

Radionuclide[a]	f_1	Radionuclide concentration (pCi/L) in drinking water yielding a risk from intake of 2 L/day for 365 days equal to that from a uniform whole body dose of: 50 mrem	170 mrem	500 mrem	5 rem
*U-234	2.0E-01	NA[b]	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA
*U-235	2.0E-01	NA	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA
*U-238	2.0E-01	NA	NA	NA	NA
	5.0E-02	NA	NA	NA	NA
	2.0E-03	NA	NA	NA	NA

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[a] Radionuclides that are naturally-occurring are designated by an *.

[b] NA = Not available.

(50, 170, 500 or 5,000 mrem) to the estimated risk from a 1-pCi ingestion of the given radionuclide, divided by the intake rate (2 L/day) and the exposure duration (1 day, 10 days or 365 days). For example, the concentration resulting in a risk equal to that from a uniform whole body dose of 50 mrem, assuming an intake of 2 L/ day for a period of one day, is computed as:

Reference Radionuclide Concentration (pCi/L) =

$$\frac{\text{risk from reference dose}}{\text{risk from 1-pCi ingestion of radionuclide}} \times \frac{\text{day}}{\text{2 L}}$$

$$\times \frac{1}{\text{exposure}} = \frac{\text{2.1 deaths in cohort from 50 mrem}}{\text{deaths in cohort from 1-pCi ingestion of the radionuclide}}$$

$$\times \frac{\text{day}}{\text{2 L}} \times \frac{1}{\text{1 day}}$$

The influence of the cohort's age at the time of acute exposure is shown in Table III-7. Estimates of the relative radiation-induced mortality in the cohort from acute ingestion of various radionuclides are presented assuming an initial cohort age of 0 years, 10 years, 20 years, 30 years and 40 years, normalized to an initial cohort age of 20 years. As one might expect, the cancer risk decreases with increasing initial age of the cohort; exposure at later ages allows less time for expression of the cancer risk. As noted previously, all estimates in Tables III-4 to III-6 assume an initial cohort age of 20 years at exposure. These estimates of risk would be

Table III-7 Relative Radiation-Induced Deaths in Cohorts of Various Beginning Ages at Exposure from Acute Ingestion of Selected Radionuclides, Normalized to Beginning Age of 20 Years

	Relative deaths in cohort				
	Cohort age at exposure (years)				
Radionuclide	0	10	20	30	40
^{90}Sr	1.2	1.1	1.0	0.9	0.7
^{137}Cs	1.5	1.3	1.0	0.8	0.6
^{210}Pb	1.4	1.2	1.0	0.8	0.6
^{226}Ra	1.3	1.2	1.0	0.8	0.3

somewhat greater for a younger beginning age and somewhat lower if a later beginning age had been assumed.

In most cases, the dominant contributors to the total mortality risk for a given radionuclide for either exposure scenario are leukemia and cancers of the G.I. tract, bone, breast, liver and kidneys. Other organs, however, may incur substantial risks in specific cases (e.g., risk to the thyroid from ingestion of radioiodines).

Some idea of the uncertainty in the preceeding dosimetric model calculations due to the choice of organ risk factors can be seen by using the ICRP 30 factors and comparing the calculations to those used in developing Table III-2. The concentrations that correspond to a dose equivalent of 4 mrem/yr for selected nuclides using these two approaches are shown in Table III-8. As can be seen in Table III-8, the differences in the calculated concentrations are generally quite small comapared to the overall uncertainty of a factor of four to five.

F. Limitations

The risk assessment presented here considers ingestion of low levels of radioactivity which could reflect conditions of environmental exposure. These estimates of potential radiation-induced cancer mortalities, however, are subject to numerous

Table III-8 Drinking water concentrations corresponding to a dose of 4 mrem/yr for selected nuclides comparing the results using two different organ risk coefficients; viz, the ICRP 30 set and the set used in the calculations for the present document based on BIER III.

Nuclide	Concentration in pCi/L corresponding to a dose of 4 mrem/yr using: ICRP 30	BEIR III/USEPA
^{3}H	89,000	88,000
^{58}Co	1600	2200
^{60}Co	210	240
^{89}Sr	680	880
^{90}Sr	41	46
^{129}I	20	14
^{131}I	100	700
^{134}Cs	74	77
^{137}Cs	110	110
^{226}Ra	5	10
^{238}U	23	82
^{239}Pu	13	39
^{241}Am	2.5	3.9

uncertainties.

Mortality rates in a real population may change significantly with time and are difficult to predict. The life table methodology as it is applied here, however, considers a hypothetical cohort with specified probabilities of death (i.e., representative of the 1970 U.S. population) which are constant throughout the period of analysis.

Additional uncertainties in estimating radiation risk with the present methodology arise from uncertainties in the dosimetric methods and risk factors. A linear dose-response relationship has been assumed for all calculations in this study. The selection of risk estimators, latency and plateau periods and other risk parameters is subject to major uncertainties, which are difficult to quantify at present.

No age dependence has been considered in the dose or risk parameters. We consider dose rates and risks to individuals exposed to a radionuclide from birth to death Sufficient information is not available to allow adequate consideration of age dependence in these calculations.

With these uncertainties in mind, the estimates of radiation-induced mortality risk computed in this analysis are presented with only one significant digit (Tables III-2 to

III-6). While it is impossible to quantify the uncertainty in these estimates in any precise manner, any greater degree of accuracy is clearly unwarranted. Several recent studies (e.g., Cuddihy et al. 1979, Dunning and Schwarz 1981, Schwarz and Dunning 1982) have indicated that estimates of organ doses from internal exposures may be associated with large uncertainties (perhaps a factor of two to ten) due to variation in the metabolic and physiological characteristics of exposed individuals. As described previously, model assumptions concerning metabolism of radioactive daughters produced in the body may also influence dose estimates significantly. Furthermore, the extrapolation from dose to health risk is potentially subject to much greater uncertainty. The accuracy of the risk estimates in this report relies, in large measure, on the validity of the linear hypothesis.

IV. ANALYTICAL METHODS[a]

A. Gross Alpha and Gross Beta Radioactivity in Drinking Water, Method 900.0

1. Scope and Application

a. This method covers the measurement of gross alpha and gross beta particle activities in drinking water. The method is a screening technique for monitoring drinking water supplies for alpha and beta particle activities according to the limits set forth under the Safe Drinking Water Act, PL 93-523, 40 FR 34324, and thereby determining the necessity for further analysis.

b. The method is applicable to the measurement of alpha emitters having energies above 3.9 MeV and beta emitters having maximum energies above 0.1 MeV.

c. The minimum limit of concentration to which this method is applicable depends on sample size, counting system characteristics, background and counting time. The National Primary Interim Drinking Water Regulations (NIPDWR) require a gross beta detection limit of 4 pCi/L, an alpha detection limit of 1 pCi/L for compliance with Part 141.15(a) and a gross alpha detection limit of 3 pCi/L for compliance with Part 141.15(b).

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[a] From Prescribed Procedures for Measurement of Radioactivity in Drinking Water, EPA 60014-80-032.

d. Since, in this method for gross alpha and gross beta measurement, the radioactivity of the sample is not separated from the solids of the sample, the solids concentration is very much a limiting factor in the sensitivity of the method for any given water sample. Also, for samples with very low concentrations of radioactivity, such as those from drinking water sources, it is essential to analyze as large a sample aliquot as is needed to give reasonable counting times in meeting the required sensitivities (detection limits) indicated above. The Regulations define sensitivity in terms of detection limits (see Part 141.25(c) of the Regulations).

e. The largest sample aliquot that should be counted for gross alpha activity is that size aliquot which gives a solids density thickness of 5 mg/cm^2 in the counting planchet. For a 2-in diameter counting planchet (20 cm^2), an aliquot containing 100 mg of dissolved solids would be the maximum aliquot size which should be evaporated and counted for gross alpha activity for that sample.

f. When the concentration of total dissolved solids (TDS) is known for a given water sample and the alpha background and the counting efficiency of a given counting system are known, the counting time that is needed to meet the required sensitivity (3 pCi/L) can be determined by equations given in Appendix C (reference at bottom of page IV-1)

g. The TDS is not as limiting for the counting of gross beta activity in a water sample as it is for gross alpha activity because beta particles are not stopped in solids as easily as are alpha particles. Very often a single sample aliquot is evaporated and counted for both gross alpha and gross beta activity. In that case, the sample aliquot size would be dictated by the solids limitations for alpha particles. For water samples that are to be counted for gross beta activity, equations in Appendix C (reference at the bottom of page IV-1) can also be used to determine the counting time necessary to meet a sensitivity for gross beta activity (4 pCi/L required by NIPDWR).

h. Radionuclides that are volatile under the sample preparation conditions of this method will not be measured. In some areas of the country, the nitrated water solids (sample evaporated with nitric acid present) will not remain at a constant weight after being dried at 105°C for two hours and then exposed to the atmosphere before and during counting. Other radioactivities may also be lost during the sample evaporation and drying at 105°C (such as some chemical forms of radioiodine). These types of water samples need to be heated to a dull red heat for a few minutes to convert the salts to oxides.

Usually, sample weights are then sufficiently stable to give consistent counting rates, and a correct counting efficiency can then be assigned. Some radioactivities, such as the cesium radioisotopes, may be lost when samples are heated to a dull red color. Such losses are limitations of the test method.

i. This method provides a rapid screening measurement to indicate whether specific analyses are required. For drinking waters with an extremely high solids content (greater than 500 ppm), Method 900.1 is recommended.

2. Summary of Method

a. An aliquot of a preserved drinking water sample is evaporated to a small volume and transferred quantitatively to a tared 2-in stainless steel counting planchet. The sample residue is dried to constant weight, reweighed to determine dry residue weight and then counted for alpha and/or beta radioactivity.

b. Counting efficiencies for both alpha and beta particle activities are selected according to the amount of sample solids from counting efficiency versus sample solids standard curves.

3. Sample Handling and Preservation

a. A representative sample must be collected from a free-flowing source of drinking water and should be large enough so that adequate aliquots can be taken to obtain the required sensitivity.

b. It is recommended that samples be preserved at the time of collection by adding enough $1\underline{N}$ NHO_3 to the sample to bring it to pH 2 (15 mL $1\underline{N}$ HNO_3/L of sample is usually sufficient). If samples are to be collected without preservation, they should be brought to the laboratory within five days and then preserved and held in the original container for a minimum of 16 hours before analysis or transfer of the sample.

c. The container choice should be plastic over glass to prevent loss due to breakage during transportation and handling.

4. Interferences

a. Moisture absorbed by the sample residue is an interference, since it obstructs counting and self-absorption characteristics. If a sample is counted in an internal proportional counter, static charge on the sample residue can cause erratic counting, thereby preventing an accurate count.

b. Non-uniformity of the sample residue in the counting planchet interferes with the accuracy and precision of the method.

c. Sample density on the planchet area should be no more than 5 mg/cm^2 for gross alpha and no more than 10 mg/cm^2 for gross beta.

d. When counting alpha and beta particle activity by a gas-flow proportional counting system, counting at the alpha plateau discriminates against beta particle activity, whereas counting at the beta plateau is sensitive to alpha particle activity present in the sample. This latter effect should be determined and compensated for during the calibration of the specific instrument being used.

5. Apparatus - See Appendix D (from reference at the bottom of page IV-1) for details and specifications.

a. Gas-flow proportional counting system or

b. Scintillation detector system.

c. Stainless steel counting planchets.

d. Electric hotplate.

e. Drying oven.

f. Drying lamp.

g. Glass desiccator.

h. Glassware.

i. Analytical balance.

6. Reagents

All chemicals should be of "reagent-grade" or equivalent whenever they are commercially available.

a. Distilled or deionized water having a resistance value between 0.5 and 2.0 mircohms/cm at 25°C.

b. Nitric acid, 1N: Mix 6.2 mL 16N HNO_3 (conc.) with deionized or distilled water and dilute to 100 mL.

7. Calibrations

a. For absolute gross alpha and gross beta measurement, the detectors must be calibrated to obtain the ratio of count rate to disintegration rate. Americium-241 (used for alpha activity in the collaborative test of this method) has a higher alpha particle energy (5.49 MeV) than those emitted by the naturally-occurring uranium and radium-226 radionuclides, but is close to the energy of the alpha particles emitted by naturally-occurring thorium-228 and radium-224. Standards should be prepared in the geometry and weight ranges to be encountered in these gross analyses. Americium-241, therefore, is the prescribed radionuclide for gross alpha calibration. The National Bureau of Standards (NBS) or NBS-traceable americium-241 is available from the Standard Reference Materials Catalog, NBS Special Publications 260, U.S. Department of Commerce (1976), and from the Quality Assurance Branch, EMSL-LV, P.O. Box 15027, Las Vegas, Nevada 89114.

b. Strontium-90 and cesium-137 have both been used quite extensively as standards for gross beta activity. Standard solutions of each of these radionuclides are readily available. Cesium is volatile at elevated temperatures (above 450°C). Some water supplies have dissolved solids (salts) that, when converted to nitrate salts, are quite hygroscopic and need to be converted to oxides by heating to red heat to obtain sample aliquots that are weight-stable. Sample weight stability is essential to gross alpha and gross beta measurements to ensure the accuracy of the self-absorption counting efficiency factor to be used for the samples. Strontium-90 in equilibrium with its daughter, yttrium-90, is the prescribed radionuclide for gross beta calibrations.

c. For each counting instrument to be used, the analyst should prepare separate alpha and beta particle self-absorption graphs showing water sample residue weight (mg) versus the efficiency factor (dpm/cpm), using standard alpha and beta emitter solutions and tap water. For the alpha graph standard, alpha activity is added to varying size aliquots of tap water, such that the aliquot residue weight is varied between 0 and 100 mg (for a 2-in counting planchet). A similar graph is prepared with standard beta activity and tap water aliquots, varying the residue weight between 0 and 300 mg (for a 2-in planchet). If it is planned to use water sample aliquot volumes that always contain 100 mg of dried water solids, then only the efficiency factor for that residue weight needs to be established.

d. Tap water aliquots with added americium-241 or strontium-90 standard should be acidified with a few mL of 16N HNO_3, evaporated to a small volume in a beaker on a hot plate, transferred quantitatively in 5-mL portions or less to a tared counting planchet, evaporated to dryness and, finally, dried at 105°C for two hours (or flamed to a red heat if dried solids appear to be noticeably hygroscopic). Weight-stable aliquot residues should then be alpha and/or beta counted until at least 10,000 total counts have been accumulated. A single set of reference standards prepared in this way can be used for each counting instrument for separate graph preparations and can be stored for reverification whenever needed.

8. Procedure

a. Transfer to a beaker an aliquot of a water sample of a volume size that contains no more than 100 mg (for alpha only or alpha and beta determination) or 200 mg (for beta only determination) of total water solids. Evaporate the aliquot to near dryness on a hot plate. If water samples are known or suspected to contain chloride salts, those chloride salts should be converted to nitrate salts before the sample residue is transferred to a stainless steel planchet (chlorides will attack stainless steel and increase the sample solids, and no correction can be made for those added solids). Chloride salts can be converted to nitrate salts by adding 5-mL portions of $16\underline{N}$ HNO_3 to the sample residue and evaporating to near dryness. (Two treatments are usually sufficient.) Add 10-mL of $1\underline{N}$ HNO_3 to the beaker and swirl to dissolve the residue. Quantitatively transfer the aliquot concentrate in small portions (not more than 5-mL at a time) to a tared planchet, evaporating each portion to dryness.

b. Dry the sample residue in a drying oven at 105°C for at least two hours, cool in a desiccator, weigh and count. Store the sample residue in a desiccator until ready for counting.

c. Some types of dissolved water solids, when converted to nitrate salts, are quite hygroscopic even after being dried at 105°C for two hours. When such hygroscopic salts are present in samples that are put into an automatic counting system, those samples gain weight while they are waiting to be counted, and inaccurate counting data result. When there is evidence of hygroscopic salts in sample counting planchets, it is recommended that they be flamed to a dull red heat with a Meeker burner for a few minutes to convert the nitrate salts to oxides before weighing and counting.

d. Count for alpha and beta activity at their respective voltage plateaus. If the sample is to be recounted for reverification, store it in a desiccator.

> Note: As long as counting chambers are capable of handling the same size planchet, alpha and beta activity can be determined at their respective voltage plateaus in the designated counting instruments. Keep the planchet in the desiccator until ready to count, because vapors from moist residue can damage the detector and window and can cause erratic measurements. Samples may be counted for beta activity immediately after drying, but alpha counting should be delayed at least 72 hours until equilibrium has occurred. If the gas-flow internal proportional counter does not discriminate for the higher energy alpha pulses at the beta plateau, the alpha activity must be subtracted from the beta plus alpha

activity. This is particularly important for samples with high alpha activity.

9. Calculations

a. Calculate the alpha radioactivity by the following equation:

$$\text{Alpha (pCi/L)} = \frac{A \times 1{,}000}{2.22 \times C \times V}$$

where:

A = net alpha count rate (gross alpha count rate minus the background count rate) at the alpha voltage plateau.
C = alpha efficiency factor, read from graph of efficiency versus mg of water solids/cm^2 of planchet area (cpm/dpm).
V = volume of sample aliquot (mL).
2.22 = conversion factor from dpm/pCi.

b. Calculate the beta radioactivity by the following equations:

(1) If there are no significant alpha counts when the sample is counted at the alpha voltage plateau, the beta activity can be determined from the following equation:

$$\text{Beta (pCi/L)} = \frac{B \times 1{,}000}{2.22 \times D \times V}$$

where:

B = net beta count rate (gross count rate minus the background count rate) at the beta voltage plateau.
D = beta efficiency factor, read from the graph of efficiency versus mg of water solids/cm^2 of planchet area (cpm/dpm).
V = volume of sample aliquot (mL).
2.22 = conversion factor from dpm/pCi.

(2) When counting beta radioactivity in the presence of alpha radioactivity by gas-flow proportional counting systems (at the beta plateau), alpha particles are also counted. Since alpha particles are more readily absorbed by increasing sample thickness than are beta particles, the alpha/beta count ratios vary with increasing sample thickness. Therefore, it is necessary to prepare a calibration curve by counting standards containing americium-241 with increasing thickness of solids on the alpha plateau and then on the beta plateau, plotting the ratios of the two counts versus density thickness. The alpha amplification factor (E) from that curve is used to correct the amplified alpha count on the beta plateau. When significant alpha activity is indicated by the sample count at the alpha voltage plateau, the beta activity of the sample can be determined by counting the sample at the beta voltage plateau and calculating the activity from the following equation:

$$\text{Beta (pCi/L)} = \frac{(B - AE) \times 1{,}000}{2.22 \times D \times V}$$

where:

B = net beta count rate (gross count rate minus the background count rate at the beta voltage plateau).
D = beta efficiency factor, read from the graph of efficiency versus mg of water solids/cm^2 of planchet area (cpm/dpm).
A = net alpha count rate (gross alpha count rate minus the background count rate) at the alpha voltage plateau.
E = alpha amplification factor, read from the graph of the ratio of alpha counted at the beta voltage/alpha counted at the alpha voltage versus sample density thickness.
V = volume of sample aliquot (mL).
2.22 = conversion factor from dpm/pCi.

c. Errors associated with the results of the analysis should also be reported (see Appendix B in the reference at the bottom of page IV-1 for error and statistical calculations).

10. Precision and Accuracy

a. In an interlaboratory collaborative test of the method, three sets of samples were analyzed by 18 laboratories for gross alpha and gross beta activity. The samples were prepared with dissolved water solids with known additions of americium-241 for gross alpha and cesium-137 for gross beta activity. Sample series A contained only americium-241 radioactivity, series B contained only cesium-137 radioactivity, and series C contained both americium-241 and cesium-137 radioactivities. Participating laboratories were supplied with standard solutions of americium-241 and cesium-137 and blank solutions of dissolved water solids for preparing sample self-absorption curves.

b. The gross alpha data from two laboratories were rejected for the statistical analysis because their scores in the ranked results of the laboratory averages were out of the acceptable range for 18 laboratories. The gross beta data from three laboratories were rejected for the statistical analysis for the same reason.

c. The coefficients of variation for the combined within-laboratory precision for gross alpha analysis of the three samples ranged from 7.4% to 12.2%. The coefficients of variation for the precision of the method between laboratories ranged from 11.5% to 14.6% for gross alpha analysis for the three samples.

d. The coefficients of variation for the combined within-laboratory precision for gross beta analysis for the three samples ranged from 3.5% to 5.2%. The coefficients of variation for the precision between laboratories for gross beta analysis for the three samples ranged from 3.5% to 7.5%. The coefficients of variation for the total error between laboratories based on a single analysis ranged from 5.9% to 8.3% for gross beta analysis of the three samples.

e. In the statistical test to detect method bias, the calculated values for "t" were well below the specified critical value for "t" for both gross alpha and gross beta analysis, indicating no bias in the method. Also, a comparison of the known values to the grand average values shows a deviation of less than 10% for alpha activity for the three samples. The same comparison for beta activity shows a deviation of less than 2% for each of the three samples.

f. Whenever the same radioisotopes are present in standards and samples, acceptable accuracy of measurement of alpha and beta activities would be expected. Whenever different radioisotopes are present in standards and samples, especially when significantly different particle energies are involved, any measurement of gross alpha and gross beta activity in the sample will only be an estimation of the true activities. Such an estimation can only serve to indicate the need for more specific analyses.

B. Radioactive Cesium in Drinking Water, Method 901.0

1. Scope and Application

a. This method covers the measurement of cesium-134 and cesium-137 in the same sample of drinking water. This technique makes it possible to ascertain whether a hazardous concentration of a specific nuclide is present when a gross beta screening analysis exceeds 15 pCi/L.

b. The limits set forth in PL 93-523, 40 FR 34324, recommend that in the case of man-made radionuclides, the limiting concentration is that which does not produce an annual dose equivalent greater than 4 mrem/year. This is calculated on the basis of a drinking water intake of 2 L/day using the 168-hour data listed in NBS Handbook 69. If several radionuclides are present, the sum of their annual dose equivalents must not exceed 4-mrem/year.

2. Summary of Method

a. Stable cesium carrier is added to a specified volume of drinking water. The cesium is collected as the phosphomolybdate in an acid medium. The precipitate is dissolved with base, and the ammonium and molybdate ions are removed by heating and calcium precipitation, respectively. The cesium is purified as the chloroplatinate for counting.

b. The final precipitate is counted by gamma ray spectrometry to determine cesium-134 (605 and 796 keV) and cesium-137 (662 keV) from the photopeaks in the resulting spectrum.

c. If the more restrictive cesium-134 is present alone, the planchet may be beta counted in a beta counter whose efficiency for measuring this nuclide has been previously determined using a suitable standard.

3. Sample Handling and Preservation - See Section 3, Method 900.0 of US EPA methods.

4. Interferences

a. If ammonium ions are not completely removed, the calcium molybdate scavenge will not be complete, and the final precipitate will have an incorrect chemical yield determination.

The presence of rubidium will also increase the weight of the chloroplatinate precipitate.

b. Counting efficiencies should be determined for the geometry and solids accumulation of the final precipitate.

5. Apparatus - See Appendix D (for reference at the bottom of page IV-1) for details and specifications.

a. Gamma ray spectrometer.

b. Gas-flow proportional counting system (low background beta, less than 3 cpm).

c. Plastic ring and disc mounts.

d. Mylar film.

e. Teflon filter holder.

f. Electric hotplate.

g. Analytical balance.

h. pH meter.

i. Drying oven.
j. Aluminum desiccator.

k. Centrifuge.

l. Glassware.

6. Reagents

a. Distilled or deionized water.

b. Ammonium phosphomolybdate (prepared reagent): Dissolve 100 g of molybdic acid (85% MoO_3) in a mixture of 240 mL of water and 140 mL 15N NH_4OH. When the solution is complete, filter and add 60 mL 16N HNO_3. Mix 400 mL 16N HNO_3 and 960 mL water. Allow both solutions to cool to room temperature. With constant stirring, add the ammonium molybdate solution to the nitric acid solution. Allow to stand for 24 hours. Filter through Whatman #42 filter paper. Discard the insoluble material.

Collect the filtrate in a 3-L beaker and heat to 50° to 55°C. Remove from heating unit. It is important that the solution not be heated above 55°C to avoid contamination of the precipitate with molybdic anhydride. Add 25 g NaH_2PO_4 dissolved in 100 mL water to the ammonium molybdate solution. Stir occasionally for 15 minutes and allow the precipitate to settle (approximately 30 minutes). Filter through Whatman #42 filter paper. Wash the precipitate with 1% potassium nitrate and finally with water. Dry the precipitate and paper at 100°C for three to four hours. Transfer the $(NH_4)_3P(Mo_3O_{10})_4$ solid to a weighing bottle and store in a desiccator.

c. Calcium chloride, 3M: Dissolve 33 g $CaCl_2$ in water and dilute to 100 mL.

d. Cesium carrier, 10 mg/mL, standardized: Dissolve 1.267 g CsCl in water and dilute to 100 mL in volumetric flask.

Standardization (in triplicate):

Pipette 1.0 mL cesium carrier into a 50-mL centrifuge tube. Add 15 mL water, 2 mL 0.1M chloroplatinic acid and 10 mL ethanol. Stir and cool in an ice bath for five to ten minutes. Filter through a tared glass-fiber filter. Wash the precipitate with 10 mL water, 10 mL 1N HCl and 10 mL ethanol. Dry to constant weight at 105°C, cool and weigh as Cs_2PtCl_6.

e. Chloroplatinic acid, 0.1M: Dissolve 5.2 g $H_2PtCl_6 \cdot 6H_2O$ in water and dilute to 100 mL.

f. Ethanol, 95%: C_2H_5OH.

g. Hydrochloric acid, 12N: HCl (conc.), sp.gr. 1.19, 37.2%.

h. Hydrochloric acid, 6N: Mix 1 volume 12N HCl (conc.) with 1 volume water.

i. Hydrochloric acid, 1N: Mix 1 volume 12N HCl (conc.) with 11 volumes water.

j. Sodium hydroxide, 6N: Dissolve 24 g NaOH in water and dilute to 100 mL.

7. Calibrations

a. The detection efficiency of the gamma ray spectrometer for measuring cesium-134 and cesium-137 must be obtained with commercially available calibrated standards. Geometric configurations comparable to the final separated sample should be prepared and counted so that the detection efficiency (cpm/dpm) can be determined.

b. If a beta counter is used in the measurement, a similar calibration is required with a standard tracer solution over a range of sample thicknesses (mg/cm^2).

8. Procedure

a. To a 1,000-mL drinking water sample, add 1.0 mL cesium carrier and 10 mL 12N HCl to make the solution 0.1N in HCl.

b. Slowly add 1 g $(NH_4)_3P(Mo_3O_{10})_4$ and stir for 30 minutes, using a magnetic stirrer. Allow precipitate to settle for at least four hours and discard supernate.

c. Slurry precipitate into a centrifuge tube. Centrifuge and discard supernate.

d. Wash precipitate with 20 mL 1$\underline{N}$ HCl and discard wash solution.

e. Dissolve precipitate by dropwise addition of 3 to 5 mL 6$\underline{N}$ NaOH. Heat over a flame for several minutes to remove ammonium ions. (Moist pH paper turns green as long as NH_3 vapors are evolved). Dilute to 20 mL with water.

f. Add 10 mL 3$\underline{M}$ $CaCl_2$ and adjust to pH 7 with 6$\underline{N}$ HCl to precipitate $CaMoO_4$. Stir, centrifuge and filter supernate through Whatman #41 filter paper (or equivalent) into a 50-mL centrifuge tube.

g. Wash the precipitate remaining in the centrifuge tube with 10 mL water. Filter through the same filter paper and combine the wash with filtrate. Discard filter paper.

h. Add 2 mL 0.1$\underline{M}$ H_2PtCl_6 and 5 mL ethanol. Cool and stir in ice bath for ten minutes.

i. Transfer with water to a tared glass-fiber filter. Wash with successive portions of water, 1N HCl and ethanol.

j. Dry to constant weight at 105°C, cool, weigh, mount and count, either gamma for cesium-134 and cesium-137 or beta for cesium-134 alone.

9. Calculations

a. Calculate the cesium concentration[a], D, in pCi/L as follows:

$$D = \frac{C}{2.22 \times EVR}$$

where:

C = net beta count rate (cpm) or net gamma count rate (cpm).
E = beta counter efficiency or gamma ray spectrometer efficiency.
V = liters of sample used.
R = fractional chemical yield.
2.22 = conversion factor from dpm/pCi.

10. Precision and Accuracy

A single laboratory replicate test by three analysts determined the precision and accuracy of this method. With a cesium-137 spiked water sample, the results of the nine replicate analyses, expressed in pCi/L, were:

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[a] Since drinking water regulations require cesium-134 analyses on quarterly composites only when the gross beta activity exceeds 15 pCi/L, the maximum decay would be less than 10%, assuming all activity was due to cesium-134 and all activity was collected within the first aliquot. Therefore, no decay correction is needed.

18,530	18,010	17,030
18,770	17,430	18,830
17,940	17,230	17,430

A one-way Analysis of Variance was performed to determine analyst variation:

Source	Degrees of Freedom	Sum Squares of Error	Mean Square of Error
Between Analyst	2	11,990	5,995
Within Analyst	6	24,799	4,133

$$\text{Calculated F (2,6)} = \frac{5{,}995}{4{,}133} = 1.45$$

Theoretical F (2,6) 95% Confidence Limit = 5.14

It can be concluded that there is no analyst variation. Therefore, precision will be estimated by combining all data:

$$\text{Precision } (1\sigma) = \left[\frac{x^2 - \frac{(\ x)^2}{N}}{N-1} \right]^{1/2}$$

$$= \left[\frac{(2.89 \times 10^9)-(2.88 \times 10^9)}{8} \right]^{1/2}$$

$$= 678$$

$$\text{Mean } \overline{X} = 17{,}910 \text{ pCi/L}$$

% Precision relative to mean = (678/17,910) 100

= 3.7%

Since counting errors have been minimized, the overall precision when applied to low activity samples is calculated to be:

Overall Precision (1σ) =

$$\left[(3.7)^2 + (\%\ \text{counting error})^2\right]^{1/2}$$

$$\%\ \text{Bias} = \frac{(17{,}910 - 15{,}540)}{15{,}540} \times 100 = 15.2$$

C. Gamma Emitting Radionuclides in Drinking Water, Method 901.1

1. Scope and Application

a. This method describes the use of gamma spectroscopy for the measurement of gamma photons emitted from radionuclides without separating them from the sample matrix. This technique makes it possible to ascertain whether a hazardous concentration of a specific gamma emitter is present in a drinking water sample.

b. The limits set forth in PL 93-523, 40 FR 34324, recommend that in the case of man-made radionuclides, the limiting concentration is that which will produce an annual dose equivalent of 4 mrem/year. This is calculated on the basis of a drinking water intake of 2 L/day, using the 168-hour data listed in NBS Handbook 69. If several radionuclides are present, the sum of their annual dose equivalents must not exceed 4 mrem/year.

c. Currently, two types of gamma detectors are widely used, namely, the thallium-activated sodium iodide crystal, NaI(Tl), and the lithium-drifted germanium detector, Ge(Li). The Ge(Li) detector does not detect gamma photons as efficiently as the NaI(Tl) detector, but its photon energy resolution is far better than that of the NaI(Tl) detector. Because of its energy resolution advantage and the availability of large active volume Ge(Li) detectors, a Ge(Li) detection system is recommended for measuring gamma emitting radionuclides in drinking water samples.

d. The method is applicable for analyzing water samples that contain radionuclides emitting gamma photons with energies ranging from about 60 to 2,000 keV. The required sensitivity of measurement for the more hazardous gamma emitters is listed in the National Interim Drinking Water Regulations, Section 141.25. For a method to be in compliance, the detection limits for photon emitters must be 1/10 of the applicable limit. The detection limits for cesium-134 and cesium-137, which are 10 and 20 pCi/L, respectively, are met by this procedure.

2. Summary of Method

a. A homogeneous aliquot of drinking water is put into a standard geometry for gamma counting. The counting efficiency for this geometry must have been determined with standard (known) radionuclide activity. Sample aliquots are counted long enough to meet the required sensitivity of measurement specified by the NIPDWR.

b. The gamma spectrum is printed out and/or stored in the appropriate computer-compatible device for data processing (calculation of sample radionuclide concentrations).

3. Sample Handling and Preservation - See Section 3, Method 900.0 of US EPA methods.

4. Interferences

a. Significant interference occurs when a sample is counted with a NaI(Tl) detector and the sample radionuclides emit gamma photons of nearly identical energies. Such interference is greatly reduced by counting the sample with a Ge(Li) detector.

b. Sample homogeneity is important to gamma count reproducibility and counting efficiency validity. When sample radionuclides are adsorbed on the walls of the counting container, the sample is no longer homogeneous. This problem can be lessened by adding 15 mL 1N HNO_3/L to the sample at collection time.

5. Apparatus - See Appendix D (of reference at the bottom of page IV-1) for details and specifications.

a. Large volume (greater than 50 cm^3) Ge(Li) detector or 4" x 4" NaI(Tl) detector.

b. Gamma ray spectrometer plus analyzer with at least 2,048 channels for Ge(Li) or 512 for NaI(Tl).

c. Standard geometry sample counting containers for either detector (1-pint cylindrical container or 4-L Marinelli polyethylene beaker).

d. Access to a computer.

6. Reagents

a. Radon-free distilled or deionized water for standard preparation and sample dilution.

b. Nitric acid, 1N: Mix 6.2 mL of 16N HNO_3 (conc.) with distilled water and dilute to 100 mL.

7. Calibration

a. A Ge(Li) detector-gamma spectrometer can be calibrated for energy resolution as follows. For this calibration, NBS or NBS-traceable standard solutions are prescribed. Adjust the analyzer amplifier "gain" and analog-to-digital converter "zero offset" to locate each photopeak in its appropriate channel. For a Ge(Li) detector system, a 0.5- or 1.0-keV per channel calibration is recommended. For a NaI(Tl) detector system, a 10- or 20-keV per channel calibration is satisfactory, since the energy resolution of this type detector is lower than that of the Ge(Li) detector.

b. For NaI(Tl), a library of radionuclide gamma energy spectra is prepared with known radionuclide-water sample concentrations at standard sample geometries; for Ge(Li), a single solution containing a mixture of fission products may be used. These standard solutions are available from NBS or the Quality Assurance Division, EMSL-LV. Counting efficiencies for the various gamma energies (photopeaks) are determined from the activity counts of the known value samples. A counting efficiency versus gamma energy curve is determined for each container geometry and for each detector that is to be used for sample analysis. Known amounts of various radionuclides that emit gamma photons with energies that are well spaced and distributed over the normal range of analysis may also be used for this calibration. These are put into each container geometry and gamma counted for a photopeak spectrum accumulation.

c. The detector efficiency, E, at a given photopeak energy for a given geometry is determined by using a known quantity or concentration (for a volume geometry) of a gamma-emitting radionuclide as follows:

$$E = \frac{C}{A \times B}$$

where:

C = net count rate (cpm) (integrated counts in the photopeak above the base line continuum divided by the counting time in minutes).
A = activity of radionuclide added to the given geometry container (dpm).
B = the gamma ray abundance of the radionuclide being measured (gammas/disintegration).

8. Procedure

a. Measure an aliquot of the drinking water sample in a standard geometry (one that has been calibrated).

b. Place the standard geometry container (with the sample aliquot) on a shielded Ge(Li) or NaI(Tl) detector and gamma count for a period of time that will meet the required sensitivity of measurement specified by the NIPDWR.

c. Print the gamma spectrum and/or store the spectrum on the appropriate computer-compatible device.

d. Calculate the radioactivity of the gamma emitters present in the sample.

9. Calculations

These calculations are for determinations using a Ge(Li) detector system. With a NaI(Tl) detector system, similar calculations can be done by a computer, using a library of radionuclide spectra and a least-squares (1,2) or matrix analysis program (3).

a. The isotopes indicated by the gamma spectrum are determined as follows:

(1) Identify all photopeak energies.

(2) Integrate the photopeak regions of the spectrum and subtract the area under the baseline continuum to determine the true photopeak area.

(3) Isotopes are identified by their appropriate photopeaks and ratios to each other when more than one gamma photon is emitted by an isotope in the sample.

b. Calculate the sample radionuclide concentrations, A, in pCi/L as follows:

$$A = \frac{C}{2.22 \times BEV}$$

where:

C = net count rate (cpm) in the peak area above baseline continuum.
B = the gamma ray abundance of the radionuclide being measured (gammas/disintegration).
E = detector efficiency (counts/gamma) for the particular photopeak energy being considered.
V = volume of sample aliquot analyzed (L).
2.22 = conversion factor from dpm/pCi.

10. Precision and Accuracy

a. The precision and accuracy of this test procedure will be determined by a separate collaborative study. A summary of the EMSL-LV cross-check and performance sample studies (six and two, respectively) for the period of August, 1978 to October, 1979, however, gives the following information about acceptable performance in the analyses of water samples for gamma emitting radionuclides by gamma spectroscopy. Some laboratories used Ge(Li) detector-gamma spectrometer systems, and others used NaI(Tl) detector-gamma spectrometer systems.

b. Six gamma emitting radionuclides were used in these studies: chromium-51, cobalt-60, zinc-65, ruthenium-106, cesium-134 and cesium-137. Samples for the August, 1978 and October, 1978 cross-check samples and the April, 1979 performance samples contained cobalt-60 and cesium-134. The February, 1979 cross-check samples contained cobalt-60, zinc-65, cesium-134 and cesium-137. The October, 1979 cross-check samples contained chromium-51, cobalt-60, cesium-134 and cesium-137. The October, 1978 performance samples contained cesium-134 and cesium-137.

c. Cesium-134 in seven studies was analyzed by an average of 46 laboratories for a 90.8% ± 11.6% average acceptable performance. Cesium-137 in five studies was analyzed by an average of 48 laboratories for an 87.7% ± 11.7% average acceptable performance. Since the radionuclide concentrations in the samples for all studies were well below the maximum allowable concentrations for drinking water, this non-destructive gamma emitting procedure to ascertain whether cesium-134 or cesium-137 is present is recommended as an alternate to Method 901.0.

D. Radioactive Iodine in Drinking Water, Method 902.0

1. Scope and Application

a. According to the NIPDWR, drinking water supplies designated as possibly contaminated by effluents from nuclear facilities are to be monitored quarterly for iodine-131. For drinking water supplies not designated as above, when the gross beta activity exceeds 50 pCi/L, the drinking water must be analyzed for specific radionuclides for the assignment of dose for people using those supplies. This method covers the measurement of radioactive iodine in a sample of drinking water.

b. The NIPDWR lists a required sensitivity of measurement of 1 pCi/L for iodine-131 in drinking water. This method provides for the separation and concentration of iodine from sufficiently large samples so that the 1-pCi/L required sensitivity can be met.

2. Summary of Method

a. Stable iodate carrier is added to an acidified sample of drinking water. The iodate is reduced to the iodide state with Na_2SO_3, and all iodine activity is precipitated as AgI. The precipitate is dissolved and purified with zinc powder and sulfuric acid and reprecipitated as PdI_2 for counting.

b. The final precipitate is counted with either a beta system or a beta/gamma coincidence scintillation system. Gamma ray spectrometry, however, will not meet the sensitivity of measurement required by NIPDWR.

3. Sample Handling and Preservation

a. See Section 3.a, Method 900.0.

b. The drinking water sample should not have acid added at time of collection. Radiochemical analysis for iodine-131 should be performed soon after receipt to minimize radioactive decay losses ($t_{1/2}$ = 8.04 days).

4. Interferences

a. The presence of stable iodide in the drinking water would interfere by indicating a higher chemical yield than would actually result.

b. Counting efficiencies should be determined for the range of precipitate weights anticipated.

5. Apparatus - See Appendix D (for reference at the bottom of page IV-1) for details and specifications.

a. Gamma spectrometer system or

b. Gas-flow proportional counting system (low background beta, less than 3 cpm) or

c. Beta/gamma coincidence scintillation system.

d. Plastic ring and disc mounts.

e. Mylar film.

f. Teflon filter holder.

g. Electric hot plate.

h. Analytical balance.

i. pH meter.

j. Drying oven.

k. Aluminum desiccator.

l. Centrifuge.

m. Glassware.

6. Reagents

a. Distilled or deionized water.

b. Ammonium hydroxide, 6N: Mix 2 volumes 15N NH_4OH (conc.) with 3 volumes water.

c. Ethanol, 95%: C_2H_5OH.

d. Hydrochloric acid, 6N: Mix 1 volume 12N HCl (conc.) with 1 volume water.

e. Iodate carrier, (IO_3^-), 10 mg/mL (as I^-) standardized: Dissolve 1.685 g KIO_3 in water and dilute to 100 mL. Store in a dark flask.

Standardization (in triplicate):

Pipette 1.0 mL potassium iodate (KIO_3) carrier into a 50-mL centrifuge tube containing 10 mL 0.4N HNO_3. Add 1 mL 1M Na_2SO_3 and stir in a water bath for five minutes. Add 1 mL 0.2M $PdCl_2$ to precipitate and coagulate PdI_2. Cool and filter through a tared fritted glass funnel (fine to medium). Dry at 105°C, cool and weigh as PdI_2.

f. Nitric acid, 16N: HNO_3 (conc.), sp. gr. 1.42, 70.4%.

g. Nitric aid, 0.4N: Mix 1 volume 16N HNO_3 (conc.) with 39 volumes of water.

h. Palladium chloride, $PdCl_2$, 5% solution in 0.6N HCl: reagent-grade.

i. Palladium chloride, 0.2M: Dissolve 3.542 g $PdCl_2$ in 100 mL 1N HCl or mix 118 mL 5% $PdCl_2$[a] with 50 mL 2N HCl.

j. Silver nitrate, 0.1M: Dissolve 1.7 g $AgNO_3$ in water and dilute to 100 mL. Store in a dark flask.

k. Sodium sulfite, 1M: Dissolve 5 g Na_2SO_3 in 40 mL water. Prepare fresh reagent every week.

l. Sulfuric acid, 2N: Mix 1 volume 36N H_2SO_4 (conc.) with 17 volumes of water.

m. Zinc powder: reagent-grade.

7. Calibrations

a. The detection efficiency of the gamma ray spectrometer must be obtained for measuring the iodine-131 photopeak (364 keV). With commercially available calibrated standards, geometric configurations are to be prepared that are comparable to the final separated sample so that cpm/dpm factors can be determined.

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[a] Palladium chloride, 5% in hydrochloric acid, is available as a reagent solution from several chemical houses, and the 4-oz size (118 mL) can be diluted directly to 0.2M by adding 50 mL 2N HCL to the stock reagent.

b. Similar calibrations are to be made for the beta and the beta/gamma coincidence counters.

8. Procedure

a. To a 1,000-mL drinking water sample, add 7.5 mL $16\underline{N}$ HNO_3 and 1.0 mL iodate (IO_3^-) carrier. Mix well.

b. Add 4 mL $1\underline{M}$ Ha_2SO_3 and stir for 30 minutes.

c. Add 20 mL $0.1\underline{M}$ $AgNO_3$, stir for one hour and allow to settle for another hour.

d. Decant and discard as much as possible of the supernate. Filter the remainder through a glass-fiber filter and discard filtrate.

e. Transfer the filter to a centrifuge tube and slurry with 10 mL water. Add 1 g zinc powder and 2 mL $2\underline{N}$ H_2SO_4 and stir frequently for at least 30 minutes.

f. Filter, with vacuum, through a fine-fritted glass funnel and collect filtrate in an Erlenmeyer flask. Use a few mL water to wash both residue and filter and add the wash to filtrate in the flask. Discard residue.

g. Add 2 mL 6N HCl to the filtrate and heat. Add 1 mL 0.2M $PdCl_2$ and digest for at least five minutes. Centrifuge and discard supernate.

h. Dissolve the precipitate in 5 mL 6N NH_4OH and heat in a boiling water bath. Filter through a glass-fiber filter and collect filtrate in a centrifuge tube. Discard filter and residue.

i. Neutralize the filtrate with 6N HCl, add 2 mL additional and heat in a water bath. Add 1 mL 0.2M $PdCl_2$ to reprecipitate PdI_2 and digest for ten minutes. Cool slightly.

j. Transfer to a tared glass-fiber filter with water. Wash successively with 5-mL portions of water and ethanol.

k. Dry to constant weight at 105°C, cool, mount and count either beta or beta/gamma coincidence.

Note: Ordinarily, the final PdI_2 precipitate is collected on a glass-fiber filter and counted in a low background beta system, where the background is less than 3 cpm and the counting efficiency is 35%. If the precipitate is collected on an 0.8-micron membrane filter, and dried for 30 minutes at 70°C, however, it is possible to count the PdI_2 in a beta/gamma coincidence scintillation system, where the background coincidence count is less than 0.1 cpm and the counting efficiency is approximately 40%.

9. Calculations

a. Calculate the iodine concentration, D, in pCi/L as follows:

$$D = \frac{C}{2.22 \times EVR \times A}$$

where:

C = net count rate (cpm) (gross count rate minus reagent background count rate).
E = counter efficiency.
V = liters of sample used.
R = fractional chemical yield.
A = decay correction for iodine-131 ($t_{1/2}$ = 8.04 days).
2.22 = conversion factor from dpm/pCi.

10. Precision and Accuracy

a. Following replicate analyses performed by three operators on aliquots containing 5,000 dpm/L, the accuracy and precision of this procedure were determined.

The accuracy of triplicate analyses on 1,000-mL aliquots was ± 4% of the expected value. The precision attained by the technicians based on three sets of triplicate analyses was ± 2%.

F. Radioactive Strontium in Drinking Water, Method 905.0

1. Scope and Application

a. This method covers the measurement of total strontium and soluble strontium-89 and strontium-90 in drinking water. Some naturally insoluble (and probably suspended) forms of strontium-89 and strontium-90 would also be measured by this method when samples of such drinking water supplies are acid-preserved before analysis.

b. The Drinking Water Regulations under the Safe Drinking Water Act set maximum contaminant concentrations for radionuclides in drinking water based on a drinking water intake of 2 L/day, using the 168-hour data listed in NBS Handbook 69. The maximum contaminant concentrations for strontium-89 and strontium-90 are 80 pCi/L and 8 pCi/L, respectively, with the critical organ being bone marrow. If other radionuclides are also present in the drinking water, the sum of their annual dose equivalents must not exceed 4 mrem/year. The Regulations also give a required sensitivity of measurement which is defined in terms of a detection limit. The required detection limits given for strontium-89 and strontium-90 are 10 pCi/L and 2 pCi/L, respectively.

2. Summary of Method

a. Stable strontium carrier is added to the drinking water sample, and strontium-89 and strontium-90 are precipitated from the solution as insoluble carbonates. Interferences from calcium and some radionuclides are removed by one or more precipitations of the strontium carrier as strontium nitrate. Barium and radium are removed as the chromate. The yttrium-90 daughter of strontium-90 is removed by a hydroxide precipitation step, and the separated combined strontium-89 and strontium-90 are counted for beta particle activity. The counting result, immediately ascertained, represents the total strontium activity (strontium-90 + strontium-89) plus an insignificant fraction of the yttrium-90 that has grown into the separated strontium-90. The yttrium-90 daughter grows in again and is then separated with stable yttrium carrier as hydroxide. Finally, it is precipitated as oxalate and beta counted. The strontium-90 concentration is determined by the yttrium-90 activity, and the strontium-89 concentration is then determined by difference.

b. Counting efficiency data must be obtained with standard strontium-89, strontium-90 and yttrium-90 activities. These data are used to make corrections, since strontium-89, strontium-90 and yttrium-90 emit beta particles with different energies.

3. Sample Handling and Preservation

a. It is recommended that samples be preserved with acid at the time of collection. For preservation, sufficient acid should be added to make the sample pH less than 2.

b. The Drinking Water Regulations allow for the options of quarterly compositing for an annual analysis or averaging the analyses of four quarterly samples. It is especially recommended to preserve composited samples.

c. It is recommended that no less than 1-L size samples be collected for analysis.

4. Interferences

a. Radioactive barium and radium will be carried down with radioactive strontium as carbonate. This method includes steps to separate strontium from barium and radium.

b. Samples that naturally contain significant amounts of stable strontium will cause errors in the recovery of the added strontium carrier. Blank samples to which no strontium carrier is added should be run to determine natural strontium content. Hard waters contain calcium, which precipitates with the strontium in the initial carbonate precipitation. If not separated, the calcium will cause errors in the recovery of the strontium carrier. Repeated precipitations with 16N HNO_3 (conc.) will eliminate this interference.

5. Apparatus

a. Low background beta counting system (less than 3-cpm background on the beta voltage plateau is recommended).

b. Centrifuge and 50-mL centrifuge tubes.

c. Drying oven.

d. Hot water bath.

e. Electric hot plate.

f. Analytical balance.

g. pH meter.

h. Desiccator, aluminum and/or glass.

i. Stainless steel planchets, 2-in diameter by 1/4-in deep.

j. Sintered-glass (fine) crucibles.

k. Plastic ring and disc mounts.

l. Mylar film.

m. Teflon filter holder.

n. Drying lamps.

o. Glassware.

6. Reagents

a. Distilled or deionized water is to be used, and all chemicals should be of "reagent-grade" or equivalent whenever they are commercially available.

b. Strontium carrier (10 mg/mL): Dissolve 24.16 g $Sr(NO_3)_2$ in water and dilute to 1 L in a volumetric flask with water. Mix thoroughly.

Standardization (in triplicate):

Carefully pipet 10.0-mL portions of the strontium carrier solution into separate 50-mL centrifuge tubes. Add 1 mL 6$\underline{N}$ NaOH and heat in a water bath. Slowly, and with stirring, add 15 mL of 2$\underline{N}$ Na_2CO_3 solution (see sodium carbonate solution below) and continue digesting for 15 to 20 minutes. Allow to cool and filter the $SrCO_3$ precipitate through a tared sintered-glass (fine) crucible. Wash the precipitate and the crucible walls with three 5-mL portions of distilled water, adjusted to pH 8 with 6N NH_4OH, and with three 5-mL portions of acetone. Dry the crucible for 30 minutes in a 105°C oven. Cool the crucibles in a desiccator and weigh.

$$\text{Strontium (mg/mL)} = \frac{\text{mg of } SrCO_3 \times 0.5935}{10 \text{ mL}}$$

c. Yttrium carrier (10 mg/mL): Dissolve 43 g $Y(NO_3)_3 \cdot 6H_2O$ in water plus 5 mL 16N HNO_3 (conc.) and dilute to 1 L in a volumetric flask with water. Mix thoroughly.

Standardization (in triplicate):

Carefully pipet 10.0-mL portions of the yttrium carrier solution into separate 50-mL centrifuge tubes. Add 30 mL saturated $(NH_4)_2C_2O_4 . H_2O$ to each centrifuge tube and stir. Digest in a hot water bath (near boiling) for 30 minutes. Cool in an ice bath. Filter the precipitate onto a Whatman #42 filter paper, then ignite in a tared crucible at 800°C for one hour to convert the oxalate to the oxide. Cool and weigh the crucible and calculate the yttrium concentration from the following equations:

$$Y/Y_2O_3 = 2 \times 88.92/225.84 = 0.7875$$

$$\text{yttrium (mg/mL)} = \frac{\text{mg of } Y_2O_3 \times 0.7875}{10 \text{ mL}}$$

d. Acetic acid, 5.8$\underline{N}$: Mix 1 volume 17.4$\underline{N}$ CH_3COOH (glacial) with 2 volumes of water.

e. Acetone, $(CH_3)_2CO$: anhydrous.

f. Ammonium acetate buffer: Dissolve 154 g $NH_4C_2H_3O_2$ in 800 mL of water, add 57 mL 17.4$\underline{N}$ CH_3COOH (glacial) and adjust the mixture to pH 5.5 using CH_3COOH or NH_4OH. Dilute to 1 L.

g. Ammonium hydroxide, 15$\underline{N}$: NH_4OH (conc.), sp. gr. 0.90, 56.6%.

h. Ammonium hydroxide, 6$\underline{N}$: Mix 2 volumes 15$\underline{N}$ NH_4OH (conc.) with 3 volumes of water.

i. Ammonium hydroxide, 0.1$\underline{N}$: Mix 1 volume 15$\underline{N}$ NH_4OH (conc.) with 150 volumes of water.

j. Ammonium oxalate, saturated: Into 100 mL boiling water, dissolve 10 g $(NH_4)_2C_2O_4 \cdot H_2O$. Cool.

k. Barium carrier, 10 mg/mL: Dissolve 19.0 g $Ba(NO_3)_2$ in water and dilute to 1 L with water.

l. Hydrochloric acid, 6*N*: Mix 1 volume 12*N* HCl (conc.) with 1 volume of water.

m. Methyl red indicator, 0.1%: Dissolve 0.1 g of methyl red in 100 mL ethanol.

n. Nitric acid, 16*N*: HNO_3 (conc.), sp. gr. 1.42, 70.4%.

o. Nitric acid, 6*N*: Mix 3 volumes 16*N* HNO_3 (conc.) with 5 volumes of water.

p. Nitric acid, 1*N*: Mix 1 volume 6*N* HNO_3 with 5 volumes water.

q. Phenolphthalein indicator, 1%: Dissolve 1 g phenolphthalein in 50 mL ethanol and add 50 mL water.

r. Sodium carbonate, 2*N*: Dissolve 124 g $Na_2CO_3 . H_2O$ (or 106 g Na_2CO_3) in water and dilute to 1 L with water.

s. Sodium chromate, 0.5*M*: Dissolve 117 g $Na_2CrO_4 . 4H_2O$ in water and dilute to 1 L with water.

t. Sodium hydroxide, 6*N*: Dissolve 240 g NaOH in water and dilute to 1 L with water.

u. Wetting agent solution: e.g., Photo-Flo, Eastman Kodak Co.

7. Calibrations

a. Counting efficiencies: Separate counting efficiencies should be determined for strontium-89 and strontium-90 using known amounts of the respective radioactive standards and 20.0 mg of strontium carrier, precipitated as carbonate and counted. A strontium-90 precipitate is prepared after separation of the yttrium-90 daughter by the following procedure. Add a known amount of strontium-90 standard, in the order of 1,000 disintegrations per minute (dpm), and 20 mg of strontium carrier to a 50-mL centrifuge tube. Add 20 mL of water and proceed as in Steps 8.i through 8.k. Then, for the yttrium-90 counting efficiency, continue with Steps 8.1 through 8.p.

b. Sources of supply: For strontium-90-yttrium-90, the National Bureau of Standards, Washington, DC, offers a standard solution (SRM 4234) as listed in their latest Catalog #260.

For strontium-89, Amersham Radiochemicals, Arlington Heights, Illinois, offers a standardized aqueous solution essentially free from strontium-90. This item is listed as SMZ.64 in their latest catalog.

Standard sources of strontium-89 and strontium-90 are also available from the Quality Assurance Division, U.S. Environmental Protection Agency, EMSL-LV.

8. Procedure

a. Transfer 1-L water sample aliquots to 2-L beakers. Add 2.0 mL each of strontium and barium carrier solution to each sample and blank beaker. Heat the samples to boiling; add 6N NaOH, while stirring, to the phenolphthalein end point (red color) and add 50 mL 2N Na_2CO_3 solution. Continue heating to near boiling for one hour with occasional stirring. Then, set the beakers aside for at least two hours, allowing the carbonate precipitate to settle.

b. Decant most of the clear supernate and discard it. With the remainder of the supernate and necessary water washes (adjusted to pH 8 with 6N NH_4OH), quantitatively transfer the precipitate to a 50-mL centrifuge tube. Centrifuge and discard the supernate. This precipitate will contain the strontium and barium carriers.

c. Dissolve the precipitate by the dropwise addition of 4 mL 16N HNO_3.

d. Add 20 mL 16N HNO_3 to the centrifuge tube, cool in an ice bath and stir. Centrifuge and discard the supernate, which will contain a significant fraction of the calcium present in the sample.

e. Add 20 mL 16$\underline{N}$ HNO_3 to the centrifuge tube, cool in an ice bath and stir. Centrifuge and discard supernate.

Note: If drinking water samples contain much calcium (hardness), it will be necessary to repeat Step 8.e.

f. Dissolve the strontium and barium nitrate precipitate in 25 mL water, add 2 drops methyl red indicator, neutralize to yellow color with 6$\underline{N}$ NH_4OH and then adjust the pH back to red color by adding 5.8$\underline{N}$ CH_3COOH dropwise.

g. Add 5 mL ammonium acetate buffer solution and heat in a hot water bath. Add, with stirring, 2 mL 0.5$\underline{M}$ Na_2CrO_4 and digest in the hot water bath for 15 minutes. Cool the reaction mixture and centrifuge. Transfer the supernate to a clean 50-mL centrifuge tube and discard the barium chromate residue.

Note: This residue can be saved if radioactive barium, radium or lead analysis is desired.

h. To the buffered chromate supernate, add 2 mL 15$\underline{N}$ NH_4OH and heat in a hot water bath. Add 5 mL 2$\underline{N}$ Na_2CO_3 solution and digest for 15 minutes. Cool, centrifuge and discard the supernate.

Note: In the next step, the strontium-89 and strontium-90 are separated from yttrium-90 with a yttrium carrier scavenge to start a specific ingrowth drinking water period and to get a separate radiostrontium count in the following steps.

i. Add a few drops of 16N HNO_3 to the carbonate precipitate, then add 25 mL water and 1 mL yttrium carrier. Add 1 drop of wetting agent solution (such as "Photo-Flo," an Eastman Kodak Company film processing product) and 5 mL 15N NH_4OH. Heat in a hot water bath for 15 minutes with occasional stirring. Centrifuge and transfer the supernate to a clean 50-mL centrifuge tube. Wash the yttrium hydroxide precipitate with 5 mL water, centrifuge and add this wash to the supernate. Note the time of this yttrium hydroxide precipitation, which marks the beginning of the yttrium-90 ingrowth period. From this point on, it is important to proceed without delay to the final separation and count of the strontium-89 and strontium-90 activity to minimize ingrowth of yttrium-90.

> Note: Concentrated NH_4OH sometimes contains CO_2 in solution, which will cause precipitation of some of the strontium carrier in this step. If low carrier recoveries are obtained in Step 8.k, then for subsequent strontium analyses, anhydrous NH_3 gas may be substituted for concentrated NH_4OH in Step 8.i by bubbling NH_3 gas in the sample solution until the phenolphthalein end point is reached and then for five minutes more. The same precaution might be taken in Step 8.n to

prevent carrydown of the strontium-90 as the carbonate precipitate in that step.

j. Add 5 mL 2N Na_2CO_3 to the supernate from Step 8.i, heat in a hot water bath for about ten minutes, centrifuge and discard the supernate.

k. Slurry the strontium carbonate precipitate with a few mL water and transfer quantitatively to a tared glass-fiber filter. Wash the precipitate with three 10-mL portions of water, adjusted to pH 8 with NH_4OH, then with three 10-mL portions of acetone. During filtration and washes of the strontium carbonate, minimize the time of air flow through the filter to avoid collection of radon daughters. Dry the filter at 105°C for ten minutes and then weigh, mount and count (within two hours). This count gives the total of strontium-89 and strontium-90 activities, plus the ingrown yttrium-90. Note the time of this count, since it must be corrected for Yttrium-90 ingrowth (time between Steps 8.i and 8.k).

Note A: An alternative to Step 8.k involves the collection and counting of the strontium carbonate precipitate on a tared stainless planchet. For this, the approach is as follows:

(1) Slurry the strontium carbonate precipitate with a few mL water and transfer quantitatively to a tared stainless steel planchet. Dry under infrared lamp.

(2) Cool, weigh and beta count (within two hours).

Note B: The calculation of the total strontium activity, D, in the sample at this point in time can be made as follows:

$$D = \frac{C}{2.22 \times EVR}$$

where:

C = net count rate (cpm).
E = counter efficiency for strontium-90.
V = liters of sample used.
R = fractional chemical yield.
2.22 = conversion factor from dpm/pCi.

Strontium-90 (by Yttrium-90)

l. After counting the strontium carbonate for strontium-89 and strontium-90 activity, store the filter or the planchet for a measured period of ingrowth and then proceed with the following steps for yttrium-90 separation. An ingrowth period of two weeks or more is recommended for samples with very low strontium-90 activity. Step 8.i was the beginning of this ingrowth period.

m. Undo the mylar covering from the nylon ring and disc and transfer the filter to a small funnel which has been placed to drain into a 50-mL centrifuge tube. Dissolve the strontium precipitate by carefully wetting the filter with 5 mL of 6$\underline{N}$ HNO_3. Wet the filter with 2.0 mL yttrium carrier. Rinse the strontium and yttrium into the centrifuge tube by washing the filter with four 5-mL portions of 1$\underline{N}$ HNO_3. Remove the funnel from the centrifuge tube, discard the filter and add 1 drop of wetting agent solution to the centrifuge tube. Swirl the tube to mix the contents thoroughly.

Note: In the case of the stainless steel planchet:

(1) After the period for yttrium-90 ingrowth, slurry the precipitate on the planchet with 2 mL water and transfer to a centrifuge tube with the aid of a rubber policeman. To make the transfer quantitative, wash the residue from the planchet with a small amount of 1N HNO_3. Dissolve the precipitate in the tube with sufficient 1N HNO_3 and dilute with water to 10 mL.

(2) Add 2.0 mL yttrium carrier and stir.

(3) Boil to expel dissolved carbon dioxide; cool to room temperature.

n. Precipitate the yttrium as hydroxide by adding 10 mL 15N NH_4OH to the centrifuge tube, stirring and heating for ten minutes in a hot water bath. Cool, centrifuge and decant supernate into a 100-mL beaker. Note time of last precipitation; this is the end of yttrium-90 ingrowth and the beginning of yttrium-90 decay.

o. Dissolve precipitate in 1 mL 1N HNO_3 and dilute with water to 10 mL.

p. Reprecipitate yttrium by dropwise addition of 15N NH_4OH.

q. Centrifuge and combine supernate with solution in the 100-mL beaker (Step 8.n).

r. Repeat Steps 8.o and 8.p. Save the combined supernates in the beaker for strontium gravimetric yield determination (Step 8.v).

Note: Steps 8.v to 8.y are a repeat of the strontium carbonate precipitation to determine chemical yield after the yttrium has been removed.

s. Dissolve the precipitate in 15 mL water containing 2 mL 6N HCl. Precipitate the yttrium as oxalate by adding 20 mL saturated $(NH_4)_2C_2O_4$ and heating for 30 minutes in a hot water bath (near boiling). Cool in an ice bath and then filter the yttrium oxalate onto a Whatman #42 filter (4.25-cm diameter). Wash the precipitate with three 5-mL portions of water and then with three 5-mL portions of acetone. Air dry the filter for about one hour.

Note: A pH of 1.7 to 1.9 in the solution from which yttrium oxalate is being precipitated is necessary to get a uniform $9H_2O$ hydrate precipitate. This is necessary if the analyst prefers and is going to weigh the yttrium oxalate for chemical yield. The analyst may then prefer to use a tared glass-fiber filter instead of a Whatman #42 paper filter. The filter plus oxalate precipitate is weighed to determine chemical yield (recovery). See note following Step 9.a for calculations. If this procedure is followed, Step 8.u can be eliminated.

t. Mount filter on a plastic ring and disc and count for yttrium-90 activity. Record the time of the counting for decay correction (time between Step 8.n and count time).

u. Undo the mylar covering and transfer the filter to a tared crucible. Ignite at 800°C for one hour in a muffle furnace to convert the oxalate to the oxide. Cool and weigh the crucible. Determine the yttrium recovery (see Section 9.a).

v. Warm the combined supernates from Step 8.n, add 5 mL 2N Na_2CO_3 and digest for ten minutes. Cool, centrifuge and discard supernate.

w. Wash the $SrCO_3$ with 15 mL water and discard wash solution.

x. Slurry with a few mL water and transfer quantitatively to a tared stainless steel planchet. Dry under infrared lamps.

y. Cool and weigh the planchet. Determine the strontium recovery (see Section 9.a).

9. Calculations

a. Chemical yields for strontium and yttrium:

$$a = \text{Yield factor for Sr} = \frac{\text{mg } SrCO_3 \text{ recovered}}{\text{mg Sr carrier added (as carbonate)}}$$

where

20.0 mg of strontium is equivalent to 33.7 mg $SrCO_3$.

$$b = \text{Yield factor for Y} = \frac{\text{mg } Y_2O_3 \text{ recovered}}{\text{mg Y carrier added (as oxide)}}$$

where

20.0 mg of yttrium is equivalent to 25.4 mg of Y_2O_3.

Note: If chemical yield is to be determined from the yttrium oxalate precipitate, the following calculations are used:

$$b = \text{Yield factor for Y} = \frac{\text{mg } Y_2(C_2O_4)_3.9H_2O}{\text{mg Y carrier added (as oxalate)}}$$

where

20.0 mg of yttrium is equivalent to 67.9 mg of yttrium oxalate, $Y_2(C_2O_4)_3.9H_2O$.

b. Calculations for activities at equilibrium conditions: Indicated cpm values are net cpm (reagent blank, including background, subtracted):

$$^{90}\text{Y dpm} = \frac{^{90}\text{Y cpm}}{\text{abefi}} = {}^{90}\text{Sr dpm}$$

$$^{90}\text{Sr cpm} = {}^{90}\text{Y dpm} \times c$$

$$\text{Total } {}^{89,90}\text{Sr cpm} = \text{total cpm } (\text{SrCO}_3) - \text{ingrown } {}^{90}\text{Y cpm}$$

$$\text{Ingrown } {}^{90}\text{Y cpm} = {}^{90}\text{Sr cpm} \times e \times g$$

$$^{89}\text{Sr cpm} = \frac{\text{total Sr cpm}}{a} - {}^{90}\text{Sr cpm} - {}^{90}\text{Sr dpm} \times e \times g$$

$$^{89}\text{Sr dpm} = \frac{^{89}\text{Sr cpm}}{d}$$

$$^{90}\text{Sr pCi/L} = \frac{^{90}\text{Sr dpm}}{2.22 \times V}$$

$$^{89}\text{Sr pCi/L} = \frac{^{89}\text{Sr dpm}}{}$$

where:

a = strontium recovery factor
b = yttrium recovery factor
c = strontium-90 counting efficiency.
d = strontium-89 counting efficiency.
e = yttrium-90 counting efficiency.
f = yttrium-90 decay factor.
g = yttrium-90 ingrowth factor, for unwanted yttrium-90 in total strontium-89, strontium-90 count.
h = strontium-89 decay factor.
V = volume of sample analyzed (in L).
i = yttrium-90 ingrowth factor for strontium-90 by yttrium-90 determination.
2.22 = conversion factor from dpm/pCi.

Error associated with the results of the analysis should also be reported. See Section 10 for error and statistical calculations, for yttrium-90 decay and ingrowth factors and for strontium-89 decay factors.

10. Calculation Factors

a. Error and statistical calculations: Because of the random nature of radioactivity disintegrations, there is an error associated with any measured count of these disintegrations. The variability of any measurement is indicated by the standard deviation. The standard deviation in the counting rate, (R), is determined by the following equation:

$$(R) = \left[\frac{R_o}{t_1} + \frac{B}{t_2} \right]^{1/2}$$

where:

R_o = gross count rate.
t_1 = counting time for the gross count.
B = background count rate.
t_2 = counting time for the background count.
Let $(R_1) = D_1$ = standard deviation for the count of total strontium-89 and strontium-90 (from $SrCO_3$ precipitate, which includes the unwanted ingrown yttrium-90).
Let $(R_2) = D_2$ = standard deviation for the yttrium-90 count for strontium-90 determination.

The counting errors, E, for a given sample for the strontium-89 and strontium-90 determinations, expressed in pCi/L, are shown as follows:

$$\text{For } {}^{90}\text{Sr, } E = \frac{1.96\ D_2 \times 1{,}000}{2.22 \times abefiV}$$

$$\text{For } {}^{89}\text{Sr, } E = \frac{1.96 \times 1{,}000}{2.22 \times adV} \left[(D_1)^2 + \left[\frac{c + e \times g}{befi} \right]^2 (D_2)^2 \right]^{1/2}$$

where:

1.96 = 95% confidence factor.
1,000 = mL/L.
2.22 = conversion factor from dpm/pCi
a = strontium recovery factor.
b = yttrium recovery factor.
c = strontium-90 counting efficiency.
d = strontium-89 counting efficiency.
e = yttrium-90 counting efficiency.
f = yttrium-90 decay factor.

g = ingrowth factor for unwanted yttrium-90 in total radiostrontium count.
i = ingrowth factor for yttrium-90 for strontium-90 determination.

These were derived by applying propagation of error theory to the expressions in Section 9.b.

The standard deviations of a number of experimental analyses or observations are determined by:

$$S = \left[\sum_{i=1}^{n} (x_i - \overline{x})^2 / n - 1 \right]^{1/2}$$

where:

x_i = activity (pCi/L) of a given sample.
$\overline{x}$ = mean activity (pCi/L) of a series of analyses.
n = the number of replicate analyses.

b. Yttrium-90 decay and ingrowth factors (0 to 71 hours)

t(hr)	e^{-} t	1-e^{-} t	t(hr)	e^{-} t	1-e^{-} t	t(hr)	e^{-} t	1-e^{-} t
0.0	1.0000	.0000	22.5	.7843	.2157	45.0	.6151	.3849
0.5	.9940	.0054	23.0	.7801	.2199	45.5	.6118	.3882
1.0	.9893	.0107	23.5	.7759	.2241	46.0	.6085	.3915
1.5	.9839	.0161	24.0	.7717	.2283	46.5	.6053	.3947
2.0	.9786	.0214	24.5	.7676	.2324	47.0	.6020	.3980
2.5	.9734	.0266	25.0	.7634	.2366	47.5	.5988	.4012
3.0	.9681	.0319	25.5	.7593	.2407	48.0	.5955	.4045
3.5	.9629	.0371	26.0	.7552	.2448	48.5	.5923	.4077
4.0	.9577	.0423	26.5	.7512	.2488	49.0	.5891	.4109
4.5	.9526	.0474	27.0	.7471	.2529	49.5	.5860	.4140
5.0	.9474	.0526	27.5	.7431	.2569	50.0	.5828	.4172
5.5	.9423	.0577	28.0	.7391	.2609	50.5	.5797	.4203
6.0	.9373	.0627	28.5	.7351	.2549	51.0	.5766	.4234
6.5	.9322	.0678	29.0	.7311	.2689	51.5	.5735	.4265
7.0	.9272	.0728	29.5	.7272	.2728	52.0	.5704	.4296
7.5	.9222	.0778	30.0	.7233	.2767	52.5	.5673	.4327
8.0	.9172	.0828	30.5	.7194	.2806	53.0	.5642	.4358
8.5	.9123	.0877	31.0	.7155	.2845	53.5	.5612	.4388
9.0	.9074	.0926	31.5	.7117	.2883	54.0	.5582	.4418

t(hr)	e^{-} t	1-e^{-} t	t(hr)	e^{-} t	1-e^{-} t	t(hr)	e^{-} t	1-e^{-} t
9.5	.9025	.0975	32.0	.7078	.2922	54.5	.5552	.4448
10.0	.8976	.1024	32.5	.7040	.2960	55.0	.5522	.4478
10.5	.8928	.1072	33.0	.7002	.2998	55.5	.5492	.4508
11.0	.8880	.1120	33.5	.6965	.3035	56.0	.5462	.4538
11.5	.8832	.1168	34.0	.6927	.3073	56.5	.5433	.4567
12.0	.8785	.1215	34.5	.6890	.3110	57.0	.5404	.4596
12.5	.8737	.1263	35.0	.6853	.3147	57.5	.5375	.4625
13.0	.8690	.1310	35.5	.6816	.3184	58.0	.5346	.4654
13.5	.8644	.1356	36.0	.6779	.3221	58.5	.5317	.4683
14.0	.8597	.1403	36.5	.6743	.3257	59.0	.5288	.4712
14.5	.8551	.1449	37.0	.6706	.3294	59.5	.5260	.4740
15.0	.8505	.1495	37.5	.6670	.3330	60.0	.5232	.4768
15.5	.8459	.1541	38.0	.6634	.3366	60.5	.5203	.4797
16.0	.8413	.1587	38.5	.6599	.3401	61.0	.5175	.4825
16.5	.8468	.1632	39.0	.6563	.3437	61.5	.5148	.4852
17.0	.8323	.1677	39.5	.5428	.3472	62.0	.5092	.4880
17.5	.8278	.1722	40.0	.6493	.3507	62.5	.5092	.4908
18.0	.8234	.1766	40.5	.6458	.3542	63.0	.5065	.4935
18.5	.8189	.1811	41.0	.6423	.3577	64.0	.5010	.4990
19.0	.8145	.1855	41.5	.6388	.3612	65.0	.4957	.5043
19.5	.8101	.1899	42.0	.6354	.3646	66.0	.4903	.5097
20.0	.8058	.1942	42.5	.6320	.3680	67.0	.4851	.5149
20.5	.8014	.1986	43.0	.6286	.3714	68.0	.4799	.5201
21.0	.7971	.2029	43.5	.6252	.3748	69.0	.4747	.5253
21.5	.7928	.2072	44.0	.6219	.3781	70.0	.4696	.5304
22.0	.7885	.2115	44.5	.6185	.3815	71.0	.4646	.5354

c. Yttrium-90 ingrowth factors (0 to 27 days)

t(days)	1-e^{-} t	t(days)	1-e^{-} t	t(days)	1-e^{-} t
0.00	.0000	9.00	.9029	18.00	.9906
0.25	.0627	9.25	.9090	18.25	.9912
0.50	.1215	9.50	.9147	18.50	.9917
0.75	.1766	9.75	.9201	18.75	.9922
1.00	.2283	10.00	.9251	19.00	.9927
1.25	.2767	10.25	.9298	19.25	.9932
1.50	.3221	10.50	.9342	19.50	.9936
1.75	.3646	10.75	.9384	19.75	.9940
2.00	.4045	11.00	.9422	20.00	.9944
2.25	.4418	11.25	.9458	20.25	.9948
2.50	.4768	11.50	.9492	20.50	.9951
2.75	.5097	11.75	.9524	20.75	.9954
3.00	.5404	12.00	.9554	21.00	.9957
3.25	.5692	12.25	.9582	21.25	.9959
3.50	.5963	12.50	.9608	21.50	.9962
3.75	.6216	12.75	.9633	21.75	.9964
4.00	.6453	13.00	.9656	22.00	.9967

t(days)	$1-e^{-\ t}$	t(days)	$1-e^{-\ t}$	t(days)	$1-e^{-\ t}$
4.25	.6676	13.25	.9678	22.25	.9969
4.50	.6884	13.50	.9697	22.50	.9971
4.75	.7080	13.75	.9716	22.75	.9973
5.00	.7263	14.00	.9734	23.00	.9974
5.25	.7435	14.25	.9751	23.25	.9976
5.50	.7596	14.50	.9766	23.50	.9977
5.75	.7746	14.75	.9781	23.75	.9979
6.00	.7888	15.00	.9795	24.00	.9980
6.25	.8020	15.25	.9808	24.25	.9981
6.50	.8145	15.50	.9820	24.50	.9982
6.75	.8261	15.75	.9831	24.75	.9984
7.00	.8370	16.00	.9842	25.00	.9985
7.25	.8472	16.25	.9852	25.25	.9986
7.50	.8568	16.50	.9861	25.50	.9987
7.75	.8658	16.75	.9870	25.75	.9987
8.00	.8742	17.00	.9878	26.00	.9988
8.25	.8820	17.25	.9886	26.25	.9989
8.50	.8896	17.50	.9893	26.50	.9990
8.75	.8864	17.75	.9900	26.75	.9990
				27.00	.9991

d. Strontium-89 decay factors (0 to 59.5 days)
($t_{1/2}$ = 51 days)

t(days)	$e^{-\ t}$	t(days)	$e^{-\ t}$	t(days)	$e^{-\ t}$
0.0	1.0000	20.0	.7620	40.0	.5808
0.5	.9932	20.5	.7569	40.5	.5769
1.0	.9865	21.0	.7518	41.0	.5730
1.5	.9798	21.5	.7568	41.5	.5690
2.0	.9732	22.0	.7416	42.0	.5652
2.5	.9668	22.5	.7366	42.5	.5613
3.0	.9601	23.0	.7317	43.0	.5575
3.5	.9536	23.5	.7267	43.5	.5539
4.0	.9471	24.0	.7218	44.0	.5500
4.5	.9407	24.5	.7169	44.5	.5462
5.0	.9344	25.0	.7120	45.0	.5427
5.5	.9280	25.5	.7072	45.5	.5380
6.0	.9217	26.0	.7023	46.0	.5352
6.5	.9155	26.5	.6977	46.5	.5318
7.0	.9093	27.0	.6930	47.0	.5280
7.5	.9031	27.5	.6882	47.5	.5245
8.0	.8970	28.0	.6836	48.0	.5210
8.5	.8909	28.5	.6790	48.5	.5175
9.0	.8849	29.0	.6742	49.0	.5140
9.5	.8789	29.5	.6699	49.5	.5105
10.0	.8729	30.0	.6651	50.0	.5070
10.5	.8670	30.5	.6608	50.5	.5035

t(days)	$e^{- t}$	t(days)	$e^{- t}$	t(days)	$e^{- t}$
11.0	.8612	31.0	.6562	51.0	.5000
11.5	.8553	31.5	.6519	51.5	.4967
12.0	.8495	32.0	.6473	52.0	.4933
12.5	.8438	32.5	.6430	52.5	.4900
13.0	.8381	33.0	.6388	53.0	.4868
13.5	.8324	33.5	.6342	53.5	.4834
14.0	.8268	34.0	.6300	54.0	.4801
14.5	.8212	34.5	.6259	54.5	.4769
15.0	.8156	35.0	.6215	55.0	.4734
15.5	.8101	35.5	.6172	55.5	.4702
16.0	.8046	36.0	.6131	56.0	.4671
16.5	.7991	36.5	.6090	56.5	.4640
17.0	.7938	37.0	.6050	57.0	.4608
17.5	.7883	37.5	.6009	57.5	.4578
18.0	.7881	38.0	.5968	58.0	.4547
18.5	.7778	38.5	.5928	58.5	.4513
19.0	.7725	39.0	.5888	59.0	.4484
19.5	.7672	39.5	.5848	59.5	.4454

11. Precision and Accuracy

a. In a single operator test of the method, two sets of five water samples containing known amounts of strontium-89 and strontium-90 were analyzed for those radionuclides. The average recovery of added strontium-90 was 95% and 94% for the two sets of samples at a precision of 3% and 5% at the 95% confidence level.

b. In a collaborative test of the method with 13 laboratories participating, three samples containing known amounts of strontium-89 and strontium-90 were analyzed (samples A, B and C).

c. The data of two laboratories for all three samples for strontium-90 were rejected because their scores in the ranked results of the laboratory averages were outside the acceptable range for 13 laboratories and three samples.

d. The coefficients of variation for the three samples ranged from 11.3% for 1,000-pCi/L concentrations to 57% for 10-pCi/L concentrations.

e. The coefficients of variation for the combined within-laboratory precision for strontium-90 in the three samples ranged from 13.6% for 1,000-pCi/L concentrations to 23% for 10-pCi/L concentrations.

f. The coefficients of variation for the precision of the method between laboratories for strontium-89 in the three samples ranged from 20% for 1,000-pCi/L concentrations to 43% for 10-pCi/L concentrations.

g. The coefficients of variation for the precision of the method between laboratories for strontium-90 in the samples ranged from 15% for 1,000-pCi/L concentrations to 44% for 10-pCi/L concentrations.

h. The coefficients of variation for the total error between laboratories based on a single analysis for strontium-

89 in the three samples ranged from 23% for 1,000-pCi/L concentrations to 71% for 10-pCi/L concentrations.

i. The coefficients of variation for the total error between laboratories based on a single analysis for strontium-90 in the three samples ranged from 17% for 1,000-pCi/L concentrations to 46% for 10-pCi/L concentrations.

j. In the statistical test to detect method bias, no significant bias was shown in the analysis of the three samples for strontium-89. In the analysis of the three samples for strontium-90, sample C (1,000-pCi/L concentration) showed a low bias, but not seriously.

k. The strontium-89 analysis of samples A, B and C deviated from the known values by the factors 1.49, 1.01 and 1.03, respectively. The strontium-90 analysis of samples A, B and C deviated from the known values by the factors 1.00, 0.899 and 0.820, respectively.

F. Tritium in Drinking Water, Method 906.0

1. Scope and Application

a. This method covers the measurement of tritium (as T_2O or HTO) in a sample of drinking water by liquid scintillation spectrometry. This technique assures the identification of tritium in drinking water at a concentration 50-fold lower than promulgated in the Safe Drinking Water Act, PL 93-523.

b. The maximum contaminant level for tritium in drinking water as given in the NIPDWR is 20,000 pCi/L. The NIPDWR list a required detection limit for tritium in drinking water of 1,000 pCi/L or 1 pCi/mL, meaning that drinking water supplies, where required, should be monitored for tritium at a sensitivity of 1 pCi/mL. The use of Equation (3) in Appendix C (for reference at the bottom of page IV-1) will determine the necessary counting time required to meet the sensitivity for drinking water monitoring.

c. By counting standard tritium and background samples at the same time as the prepared drinking water samples, the results and the behavior of the liquid scintillation spectrometer can be routinely monitored.

2. Summary of Method

a. A 100-mL aliquot of a drinking water sample is treated with a small amount of sodium hydroxide and potassium permanganate. It is then distilled, and a specified fraction of the distillate is collected for tritium analysis. The alkaline treatment prevents other radionuclides, such as radioiodine and radiocarbon, from distilling over with the tritium. Some drinking water supplies will contain trace quantities of organic compounds (especially surface water sources that contain fish and other life). The permanganate treatment oxidizes trace organics in the sample aliquot which could distill over and cause quenching interferences. A middle fraction of the distillate is collected for tritium analysis because the early and late fractions are more apt to contain interfering materials for the liquid scintillation counting process.

b. The collected distillate fraction is thoroughly mixed, and a portion is mixed with liquid scintillator solution. After dark adapting, it is counted in the liquid scintillation counting system for tritium beta particle activity.

c. The scintillator solution, mixed with the radioactive sample, is excited by beta particles and emits light pulses by a molecular de-excitation process. The number of pulses per unit time is proportional to the quantity of activity present. Multiple solutes are used in the scintillator to provide the best combination of wavelength and pulse height for this application. The pulses are detected by two photomultiplier tubes connected in coincidence and converted to electric signals. The amplified pulses are recorded, and the count rate is measured. The efficiency of the system can be determined by use of prepared tritiated water standards having the same density and color as the sample.

3. Sample Handling and Preservation

The drinking water sample should be collected in its natural state and should not be acidified. Since tritium in drinking water is very much apt to be in the form of T_2O or HTO, there is no need for special handling or preservation.

4. Interferences

a. Tritium in background water is an interference. Slightly elevated levels are present in surface waters, so deep well sources for background water should be used.

b. All fluors should be checked for excitation under lighting conditions being used, and, if necessary, they should be exposed only to red light. Dioxane-base scintillators exposed to fluorescent lighting should be dark-adapted for 24 hours. Toluene- or xylene-base scintillators exposed to fluorescent lighting should be dark-adapted for a minimum of one hour.

c. The use of plastic vials may cause build-up of static charge and give erratic results.

5. Apparatus - See Appendix D (for reference at the bottom of page IIV-1) for details and specifications.

a. Coincidence-type liquid scintillation spectrometer.

b. Liquid scintillation vials: Low-potassium glass is recommended. Polyethylene vials may be used when dioxane liquid scintillator solution is used.

c. Distillation apparatus: For aqueous distillation: 250-mL and 1,000-mL round bottom Pyrex flasks, connecting side arm adapter (such as Corning part #9060), condenser, graduated cylinder, boiling chips and heating mantle.

6. Reagents

a. Reagents for distillation treatment: sodium hydroxide pellets and potassium permanganate (ACS, reagent-grade).

b. Background water with tritium activity below the minimum detectable activity (most deep well waters are low in tritium content).

c. Scintillator solutions:

(1) Solution G liquid scintillator solution: Dissolve 18 g scintillation-grade PPO (2,5-diphenyloxazole) and 3.6 g scintillation-grade BIS-MSB p-bis (o-methylstyryl) benzene in 2 L of spectroquality p-xylene. Add 1 L Triton N-101 detergent (Rohm & Haas) to the p-xylene scintillator solution. Dissolve

50 g SXS (sodium xylene sulfonate) in 100 mL distilled water and add this solution to the p-xylene scintillator-Triton solution. Mix thoroughly. Store the solution in a dark (amber) bottle. The organic solvent evaporates slowly through the wall of the polyethylene vials. For this reason, counting of samples should be completed within three days after preparation.

(2) Detergent-type liquid scintillator solutions are available as commercial preparations. They are also prepared with aromatic hydrocarbon solvents and should, therefore, when used with plastic vials, be counted within three days for the reason stated above.

(3) Dioxane liquid scintillator solution: Dissolve 4 g scintillation-grade PPO (2,5-diphenyloxazole), 0.05 g scintillation-grade POPOP 1,4-bis (5-phenyloxazolyl-2-yl) benzene and 120 g naphthalene in 1 L of spectroquality 1,4-dioxane. Store the solution in a dark (amber) bottle. This solution can be used with glass or polyethylene vials.

7. Calibrations

Determination of recovery and counting efficiency factors: See calculations, Sections 9.b and 9.c.

a. Into a 1-L volumetric flask, pipette a tritium standard solution containing approximately 1,000 disintegrations per minute (dpm)/mL and dilute to volume using low level tritium background raw water (undistilled) and standard tritium activity. Label this solution "Raw Water Tritium Standard Solution." For tritium background determinations, distill approximately 600 mL of water, obtained from the same raw water source as above (without tritium activity added). Using the distillate and standard tritium activity, prepare a tritium standard solution in a 500-mL volumetric flask containing the same activity as the "Raw Water Tritium Standard Solution." Label this solution "Distilled Water Tritium Standard Solution."

b. Aqueous permanganate distillation: To a 100-mL aliquot of the "Raw Water Tritium Standard Solution" in a 250-mL distillation flask, add 0.5 g sodium hydroxide, 0.1 g potassium permanganate and a boiling chip. Proceed according to the procedure described in Section 8.a. Discard the first 10 mL and collect 50 mL of the distillate for analysis. Mix well. Repeat the distillation with two more 100-mL aliquots for triplicate analyses.

c. For liquid scintillation counting: Prepare three aliquots of the "Raw Water Tritium Standard Solution" distillate (from Steps 7.a and 7.b), three aliquots of the "Distilled Water Tritium Standard Solution" and three aliquots of the distilled raw water (for background). Mix 4 mL water with 16 mL of the dioxane scintillator solution or 8 mL water with 12 mL of a detergent-type scintillator solution in a liquid scintillator vial (glass vials should be used for detergent-type scintillator solutions). Shake well and dark-adapt the vials overnight. Count each vial in a liquid scintillation counter long enough to meet the required detection limit (1 pCi/mL) or longer.

8. Procedure

a. Add 0.5 g sodium hydroxide and 0.1 g potassium permanganate to a 100-mL aliquot of the sample in a 250-mL distillation flask. Add a boiling chip to the flask. Connect a side arm adapter and a condenser to the outlet of the flask and insert a graduated cylinder at the outlet of the condenser (Figure IV-1). Heat the sample to boiling to distill and discard the first 10 mL of distillate as a separate fraction. (It is important that the first 10-mL fraction for samples and standards alike be discarded, since there is a gradient in the tritium concentration of the distillate.) Collect the next 50 mL of distillate for tritium analysis and mix thoroughly.

b. Mix 4 mL of the distillate with 16 mL of the dioxane liquid scintillator or 8 mL of the distillate with 12 mL of a

Figure IV-1

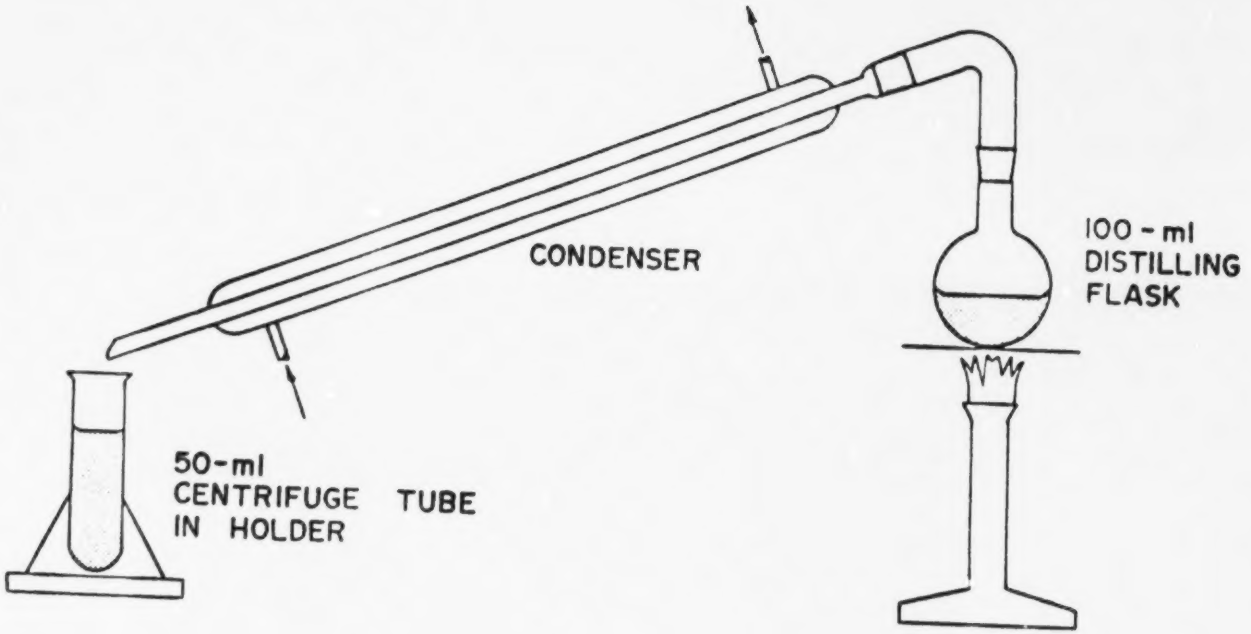
CONDENSER
100 - ml
DISTILLING
FLASK
50-ml
CENTRIFUGE TUBE
IN HOLDER

FIGURE IV-1 DISTILLATION APPARATUS FOR TRITIUM ANALYSIS

detergent-type scintillator solution in a liquid scintillation vial.

c. Prepare background and standard tritium water solutions for counting, using the same amount of water and the same scintillator as used in the preparation of samples. Use low tritium background distilled water for these preparations (distillate of most deep well water sources is acceptable, but each source should be checked for tritium activity before using).

d. Dark-adapt all samples, backgrounds and standards. Count the samples, backgrounds and standards at least long enough to meet the required detection limit (1 pCi/mL) for the sample.

Note: In normal counting operation, tritium is counted with a window setting where the figure of merit is at maximum.

$$\text{Figure of Merit} = \frac{E^2}{B} = \frac{(\text{Efficiency})^2}{(\text{Background})}$$

9. Calculations

a. Calculate the tritium concentration, A, in pCi/L as follows:

$$A = \frac{(C - B) \times 1{,}000}{2.22 \times E \times V \times F}$$

where:

C = sample count rate (cpm).
B = background count rate (cpm).
E = counting efficiency, as determined in Section 9.b.
V = volume of the sample aliquot (mL).
F = recovery factor, as determined in Section 9.c.
2.22 = conversion factor for dpm/pCi.

b. Determine the counting efficiency, E, as follows:

$$E = \frac{D - B}{G}$$

where:

D = distilled water standard count rate (cpm).
B = background count rate (cpm).
G = activity of distilled water standard (dpm).

c. Calculate the recovery correction factor, F, as follows:

$$F = \frac{L - B}{E \times M}$$

where:

L = raw water standard distillate count rate (cpm).
B = background count rate (cpm).
E = counting efficiency, as determined in Section 9.b.
M = activity of raw water standard
(before distillation) (dpm).

10. Precision

a. In an interlaboratory collaborative test of the method, three water samples were analyzed for tritium by 25 laboratories. The three water samples were prepared by spiking tap water with measured amounts of tritiated water of known concentrations.

b. The data from three laboratories for the three water samples were rejected from the statistical analysis because their scores in the ranked results of the laboratory averages were outside the acceptable range for 25 laboratories and three samples.

c. The coefficients of variation for the combined within-laboratory precision for the three samples ranged from 2.1% to 5.2%.

d. The coefficients of variation for the precision of the method between laboratories for the three samples ranged from 21.0% to 28.8%.

e. The coefficients of variation for the total error between laboratories based on a single analysis for the three samples ranged from 21.3% to 29.2%.

f. In the statistical test to detect method bias, the method showed no bias for tritium concentrations at the 7-pCi/mL level to a small bias on the low side for concentrations at the 300-pCi/mL level (average analytical value about 9% lower than the known value).

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